

Summary:

I raise concerns, supported by published research, on the following topics:

1. Loss of agricultural land and food security;
2. South Norfolk rural landscape and environment as national cultural heritage;
3. Threat to biodiversity;
4. Fire hazard from and to the development;
5. Local need for energy and domestic roof-top production;
6. Renewable energy, net zero and the various national interests (with residents as members of the nation);
7. Location of renewable energy production; roles of the local rural landscape; different approach to energy.

I am a farmer's daughter and (almost) lifelong resident of the Great Moulton area. I have studied and worked elsewhere in the UK and abroad but this is where my heart lies. I heard many well-informed and very personal statements at the Open Hearing at Dunston Hall on 15 September 2026, at which I too made a presentation. There we were only allowed 3 minutes to voice our concerns, but were encouraged to submit a more developed version to the Examining Authority, with supporting documentation. I am using an evidence-based approach to frame my concerns using scientific research, national and local government publications, specialist reports and national news reporting.

I am worried about the loss of prime agricultural land at a time of international political and climate uncertainty, when the nation's food security (in the news in the week before the Dunston Hall hearing; see also Cole and Petrikova 2024) should be of utmost importance. Norfolk is an important contributor to the UK's food security and the international food chain (Norfolk and Suffolk Agri-Food Industry Council 2022). The use of 'best and most versatile' (BMV) agricultural land should be avoided for renewable energy developments according to Paragraph 2.10.21 of the *National Policy Statement for Renewable Energy Infrastructure (EN-3)* (Department of Energy Security and Net Zero 2025). The same paragraph gives clear guidance that 'previously developed land, brownfield land, contaminated land and industrial land' should be prioritised, followed by the lowest-grade agricultural land. According to the *Predictive Agricultural Land Classification Map for England* (Department for Environment, Food and Rural Affairs 2026) the proposed site would be on BMV land. Removing this land from agricultural production can be expected to have a significant effect on food security (see Yawson *et al.* 2017; 2020) and it makes no sense to compromise high-grade agricultural land when brownfield sites and industrial roof-tops are available.

The iconic rural landscape and environmental system of south Norfolk are part of our national cultural heritage, falling within the remit of the government's 25 year environmental plan target of 'enhancing beauty, heritage and engagement with the natural environment' (Department for Environment, Food & Rural Affairs 2023). Norfolk is a source not only of agricultural revenue (£2.3bn per annum: Rural Policy Group 2026: 13) but of income from tourism (£2.8bn per annum: Norfolk County Council 2026). It is an area of fields, hedgerows and stands of trees, narrow country lanes and historic villages in an organisation rooted in its agricultural history. Road-widening and other interventions to accommodate heavy transport vehicles will change the character of the area forever and, in some cases, threaten historic buildings.

This archetypal corner of 'England's green and pleasant land' contributes to the wellbeing of both visitors and residents (see Alcock *et al.* 2015) — the value of green spaces to mental health brought into sharp perspective by the Covid Pandemic (see, for example: Hubbard *et al.* 2021; Lee *et al.* 2023). People choose to live (and visit) here because of the quiet, rural setting — enjoying its agricultural and natural rhythms. Indeed, the byline on the *Visit Norfolk* website reads 'Come as you are, leave renewed' (Visit Norfolk 2026). Nowhere on this or any other tourism-related website that I visited in preparing this statement were there images of locals or visitors 'enjoying' a huge solar

factory. Make no mistake, this is no farm, it is an industrial development that, if it goes ahead, will have replaced actual farming.

Damage to the environment and its ecosystems during the construction and functioning of the development will affect biodiversity, in direct opposition to the government's *Blueprint for Halting and Reversing Biodiversity Loss* (Department of Agriculture, Environment and Rural Affairs *et al.* 2025). How can this be OK? UK national targets 1 (protection of biodiversity and land-use change), 3 (protection of water systems), 4 (reduction of extinction risk), 8 (minimising impact of climate change on biodiversity), 9 (sustainable management and use of wild species), 10 (sustainable agriculture and food security) 12 (access to green space) and 14 (integration of biodiversity into government decision-making), summarised in Annex A (no author no date) to this document are directly relevant to this development. That is eight out of 21 targets that would be adversely affected.

Biodiversity in the UK is already very low — it ranks in the bottom 10% in the world and last among the G7 nations (de Zylva 2019/2026; Wingate 2021; Friends of the Earth 2026). Can we afford to stress it further? According to scientific research (Mitchell *et al.* 2019; National Environmental Research Council 2026; also presented on BBC *Springwatch*) a single mature oak tree (and also other tree species) can host thousands of species of birds, mammals, fungus and insects — waterways, hedgerows and verges can be expected to be similarly species-rich. Birds, mammals and insects currently move freely throughout the local landscape for food, shelter and breeding, relying on connectivity and accessibility. We cannot afford to compromise them by blocking their pathways with high fencing and flooding the enclosed spaces (and their surrounds) with bright light.

The fire risk is acutely disturbing given that this year's hot summer and wildfires may become the norm (Arnell *et al.* 2021; Perry *et al.* 2022; Tasker and Wentworth 2024). Fires at RSBP Minsmere and Dunwich Heath (Jones 2026) were devastating and terrifying — could the same thing happen here? Is the agricultural land now a threat to the solar installations? This may well be the case, Tasker and Wentworth (2024: 6) stating that:

Vulnerability of arable agricultural land may increase under climate change, with recent studies projecting more frequent dangerous fire weather in summer, when ignitions are highest and crops are dry awaiting harvesting.

I have no faith in the the proposal's meagre water-based fire-fighting provision or that it could be deployed in time to be in any way effective. Is water the correct way to fight electrical fire — can it realistically be tackled at all? Where does the water go and will it contaminate soil and water systems? Will this compromise existing ecosystems and prevent the land returning to cultivation (which should be part of the proposal) after the site is decommissioned.

I am concerned that there is no local need for this electricity. What happens if supply outstrips carrying capacity? After losing prime agricultural land, precious ecosystems and compromising our heritage landscape and wellbeing, will the company be paid to *not* generate electricity? I am also very aware that many local residents (including my family) have their own solar panels. Coincidentally, it is now time for my household to renew its contract with the energy company and I notice that the contract period has changed from two years to one and the export tariff has decreased. However, in addition to a daily standing charge, we are paying approximately twice the export tariff to import (our own?) electricity when our solar panels are not generating.

These multiple wins for the electricity company are symptomatic of energy in the UK being big business, cashing in on the government's scattershot approach in its dash towards an ever more distant (now 2050 not 2030) net zero target. A recent government turnaround on natural gas extraction in the Jackdaw gas field and possibly the Rosebank oil field (previous approval of which was overturned as unlawful: see Jack 2026; Stacey 2026) confirms why we should not trust long-term policies and glossy plans for the environment. This new approval, should it go ahead, is direct

opposition to the government's statement (Department for Energy Security and Net Zero 2025: 35–36) that:

Our plan for future oil and gas production needs to meet both our obligations to today's workers as well as our climate obligations to future generations. We are determined to meet our commitment to manage existing oil and gas fields for their lifetime and not to issue new oil and gas licenses to explore new fields. Our commitment not to issue new licenses to explore new fields sends a clear signal of the government's commitment to the global 1.5oC goal, and 35 shows the world that you can take bold but responsible steps to move away from fossil fuel production.

We are told (Department for Energy Security and Net Zero *et al.* 2025) that renewable energy and net zero are in the national interest. I agree with the then-Prime Minister, Sir Keir Starmer, that 'Energy security is national security'. However, food security, biodiversity and the nation's well-being are also in the national interest. My research has shown me that there are government policies and guidelines in place that should be protecting all these national interests and yet I have no confidence that this is happening in general, or will come into play in the specific case of the East Pye Solar Factory. Those of us living in south Norfolk, who will be directly and indirectly adversely affected by this development, are people of this nation. Our interests should count, our local knowledge valued and our concerns taken seriously. I do not see any positive outcomes for us from this development, especially when it is seen in the wider context of the proliferation of industrialised energy production (and data storage) over a huge swathe of the county. These should be considered together because their cumulative impact represents an extreme change of land use within the region.

This is the wrong development in the wrong place, according to evidence presented by local authorities, local residents, environmental and government agencies as well as according to the government's own policies and guidance. Renewable energy generation on this scale should be located on brownfield sites and on the roofs of warehouses, supermarkets and other large buildings close to where the electricity is needed. The rural landscape already has many important roles, especially in contributing to Britain's food security, cultural heritage, economy and well-being. It is not a blank canvas ripe for filling with electrical installations, AI storage and endless new housing. I foresee a future, not too far away, in which it is industrialised beyond recognition, sacrificing food security, biodiversity, essential ecosystems and everyone's fragile well-being to serve consumption elsewhere. As global warming and international politics begin to have a local impact, now is the time to take measures to protect agriculture (national food security and local economy) and the environment (biodiversity and well-being), and to find ways to *cut* energy consumption (energy security and net zero).

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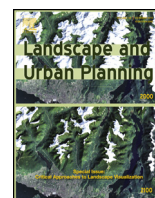
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Research Paper

What accounts for 'England's green and pleasant land'? A panel data analysis of mental health and land cover types in rural England



William Blake's phrase "England's green and pleasant land" has become a byword for those aspects of the English countryside which are idealised in the national psyche. Although the rural landscape of today is very different from the

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HIGHLIGHTS

- Rural land cover associations with mental health estimated from hybrid models.
- Intra-rural change to more natural space associated with mental health change.
- Land cover types differentially associated with mental health within individuals.
- Farmland, uplands and coastal associated with good mental health within individuals.
- Broadleaf, grassland, coastal associated with reduced odds of psychiatric caseness.

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ABSTRACT

Exposure to green space is associated with a variety of positive health states. Research to date has focused primarily on 'generic' green space in urban areas, where green space is relatively scarce and where it is dominated by playing fields and parks. The current research adds to our understanding with an examination of relationships between different types of green space and mental health in rural areas in England (approximate rural population = 4 million). The aggregate land cover classes of Land Cover Map 2007 were linked to rural residential areas (Lower-level Super Output Areas) and then linked to rural participants ($n = 2020$) in the 18-year longitudinal British Household Panel Survey. Random effects regression of mental health (as measured by GHQ12 scores) against land cover enabled effects to be simultaneously estimated from both mean between-individual differences and from within-individual differences over time. The nine natural land cover classes (Broadleaved woodland; Coniferous woodland; Arable; Improved grassland; Semi-natural grassland; Mountain, heath and bog; Saltwater; Freshwater; Coastal) were not significantly associated with differences in mental health between individuals. However, significant relationships were observed between some types of land cover and within-individual change in mental health amongst individuals who relocated during the 18 annual waves of the panel. These findings indicate the presence of important health related ecosystem services from different land cover types that have not previously been investigated and which help more effective spatial planning and land use management.

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1. Introduction

1.1. Overview

William Blake's phrase "England's green and pleasant land" (Blake, 1804) has become a byword for those aspects of the English countryside which are idealised in the national psyche. Although the rural landscape of today is very different from the

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“pleasant pastures” and “mountains green” which Blake explicitly linked to spiritual well-being at the dawn of the industrial age, the idea that physical landscape is intimately involved in well-being persists, and is increasingly the subject of multi-disciplinary empirical research (Hartig, Mitchell, de Vries, & Frumkin, 2014).

Exposure to the natural environment has been associated with better self-reported general health (Maas, Verheij, Groenewegen, de Vries, & Spreeuwenberg, 2006; Mitchell & Popham, 2007), lower prevalence of diagnosed morbidities (Maas et al., 2009), increased longevity (Takano, Nakamura, & Watanabe, 2002), less premature mortality (Mitchell & Popham, 2008) more rapid recovery from illness (Ulrich, 1984), higher levels of psychological well-being (Kaplan & Kaplan, 1989; White, Alcock, Wheeler, & Depledge, 2013a), and lower levels of anxiety and depression (Beyer et al., 2014; de Vries, Verheij, Groenewegen, & Spreeuwenberg, 2003; Maas et al., 2009; van den Berg, Maas, Verheij, & Groenewegen, 2010). Moreover, the research is starting to inform the development of tangible health promotion strategies and practices (St Leger, 2003). However, to date, much of the evidence of a relationship between natural environments and mental health and well-being has focused on urban rather than rural communities (Alcock, White, Wheeler, Fleming, & Depledge, 2014; Astell-Burt, Mitchell, & Hartig, 2014). The aim of the current research was to begin to redress this balance.

The issue of rural mental health is important because a significant minority of people live in rural residential areas. In England, around 4.2 million people live in areas classified as rural (8% of the population). Despite a long-term trend towards urbanisation, data compiled by the UN Department of Economic and Social Affairs suggests there are currently close to 200 million people living in rural areas in Europe. Whilst there are suggestions that the prevalence of common mental disorders such as anxiety and depression are lower in rural than urban areas (Weich, Twigg, & Lewis, 2006), there are substantial differences in their prevalence across English villages which are not accounted for by variation between individuals or by area levels of socio-economic deprivation (Riva, Curtis, Gauvin, & Fagg, 2009). This suggests that other aspects of rural areas, potentially including the nature of the physical environment, may be related to mental health. Furthermore, the treatment of poor mental health in rural communities may be more challenging due to geographical barriers, social isolation and to rural cultural beliefs on help-seeking for mental illness. These factors may partially explain why suicide rates also tend to be higher in some rural areas (Gunnell et al., 2012). Against this backdrop, a better understanding of factors which may support more positive mental health in rural areas is clearly desirable.

As to why previous research into the relationships between natural environments and mental health has been neglected in rural areas, it is likely to be, in part, because the focus has been on the *quantity* of ‘available green space’, which is arguably more important for built-up urban areas where green spaces may be scarce, than in rural areas where it is abundant. Previous epidemiological research, for instance, has generally dichotomised land cover as green/not green (Jorgensen & Gobster, 2010) and there is awareness that this dichotomy is particularly limited for rural areas because there is “too little variation in the quantity of green space” (Ord, Mitchell, & Pearce, 2013) between rural areas for this comparison to often be meaningful (see also Astell-Burt et al., 2014). However, although green spaces in towns and cities are generally dominated by parks, what constitutes green space in rural areas tends to be more mixed (e.g. farmland, moorland) and as yet we know relatively little about the importance of this variation in green space in rural settings. In other words, type and quality, rather than just quantity, may be particularly relevant for mental health in rural settings.

1.2. Current Research

It is suggested that results from landscape preference studies, where work with photographs gives evidence of preferences for more natural landscapes over more urban landscapes, may be related to the potential for natural environments to reduce stress and improve well-being (Hartig & Evans, 1993; van den Berg, Koole, & van der Wulp, 2003). Drawing on insights from the landscape preferences literature on differential preferences for different land cover types, we hypothesised that the presence of certain types of natural space environment in rural areas, such as woodlands and aquatic environments, may be linked to more positive mental health outcomes. These hypotheses were tested using data from a sample of residents of rural England who took part in a longitudinal panel survey between 1991 and 2008. Using data of this kind allowed us to do two things. First, similar to the cross-sectional approach used in most previous work looking at green space and mental health in urban areas, we were able to compare the individual average mental-health of people who lived in one type of rural area (e.g. with high proportions of woodland, or of arable) with that of those living in a different type of rural area (e.g. with low proportions of woodland, or of arable). Second, due to the longitudinal nature of the data we were able to track people's well-being over many years, including among those who moved from one rural area to another rural area. Controlling for other changes in their lives, therefore, we were able to estimate the effect of different rural land cover on the same individuals over time. This second type of analysis enables us to account for factors such as personality and early life experiences. Specifically, cross-sectional correlations may merely reflect the fact that different sorts of people, with better mental health, have greater exposure to natural environments, rather than point to a causal relationship. In contrast, associations based on within-individual differences eliminate confounding from individual level heterogeneity (Antonakis, Bendahan, Jacquart, & Lalive, 2010).

By examining the relationships between mental health and a variety of land cover types in different rural residential areas, our work has parallels to pioneering research conducted in Sweden using the Scania Green Score (SGS). The SGS identified a set of characteristics which Swedish people regarded as important in green space and operationalised these using criteria measurable from land cover, land use and topography datasets (Skärbäck et al., 2012). SGS was found to be related to neighbourhood satisfaction, physical activity, Body Mass Index, vitality in women and self-rated health (Björk et al., 2008; de Jong, Albin, Skärbäck, Grahn, & Björk, 2012). A further study showed that interaction between some of the SGS component green space characteristics and being physically active was associated with reduced risk of poor mental health in women (Annerstedt et al., 2012). However, although the work from Sweden used land cover in the operationalisation of the SGS, the associations observed between desirable green space characteristics and health are difficult to interpret in simple land cover terms since single land parcels may contribute varying weight to the summative SGS ordinal measure (for further details see Skärbäck et al., 2012). Furthermore, the SGS takes account only of the presence/absence of land parcels with desirable characteristics, and not how much or little of a residential area these comprise. The current study builds upon this approach by accounting for the amount as well as the presence of different types of green space environment (although limited to land cover measures only).

The theme of our research was thus whether mental health is better in rural areas with greater amounts of certain types of land cover, and we had three specific research questions. First, we explored whether the positive association found between mental health (General Health Questionnaire) and the quantity of green space (compared to built-up areas) in urban areas would extend to

rural areas. The measure used for this includes Saltwater, Freshwater and Coastal land cover in addition to types of vegetation, and therefore we refer to it as 'natural space' rather than 'green space'. Second, and perhaps most importantly, we investigated whether the make-up of green space in rural areas would be related to mental health. For instance, compared to built-up land, would the amount of woodland (O'Brien & Morris, 2013) or coastal margins (White, Alcock, Wheeler, & Depledge, 2013b) be more positively related to mental health than other land cover types such as arable land or saltwater estuary? Our work was exploratory, in the sense that we began with the idea that different types of rural land cover may have different relationships to mental health, perhaps due to their different amenity, aesthetic or biodiversity characteristics, rather than with more specific directional hypotheses. Finally, a robustness check on these outcomes was conducted to see whether the operationalisation of the GHQ measure, as an interval scale or 'caseness' score (see below), might influence results.

2. Methods

Measures of exposure to classes of natural space were derived using GIS to link the land parcel classification of Land Cover Map 2007 (Morton et al., 2011) to English rural residential areas. Natural land cover types were expressed as a proportion of total residential area land cover. British Household Panel Survey (BHPS) data collected between 1991 and 2008 (Taylor, Brice, Buck, & Prentice-Lane, 2010) were used to derive estimates of associations between the land cover characteristics of residential areas and mental health as measured by the General Health Questionnaire (GHQ) for people living in rural areas.

2.1. Sample

The sample consisted of 2020 individuals drawn from the larger BHPS who: (a) were resident in English rural neighbourhoods, i.e. areas classified by morphology as 'Village and Dispersed', as opposed to either 'Small Town and Fringe' or 'Urban' by the Office for National Statistics in 2005 using the methodology described by Bibby and Shepherd (2004); and (b) had full data on the outcome and predictor variables, for at least one wave. Due to the longitudinal nature of the study, individuals were repeatedly interviewed over several years so each individual could have multiple 'observations'. In the sample we used here there were 12,697 observations pertaining to 2,020 individuals (mean number of observations per individual = 6.3). It is these observations that are used in subsequent statistical analyses and enable us to simultaneously compare results both between different individuals and within the same individuals over time. The observations used in the estimation amount to 61.6% of a total 20,600 BHPS observations of individuals in rural residential areas of England. Observations where covariate data were missing were excluded from analyses.

Rural status was derived from the classification of the Lower-level Super Output Area (LSOA; boundaries defined in 2001) in which individuals lived. The rural status designated in 2005 was carried across to all waves of data 1991–2008; it is unlikely that any areas are misclassified as rural since that would require either an area previously classified as a small town or as urban to have become a village by the time of the 2005 classification, or for the morphology of a village to have changed immediately after 2005. However, it is possible that a small number of areas were rural at earlier times in the panel but are excluded from the analysis by this decision rule, as villages grew into small towns by 2005.

LSOAs are a standard residential area unit used in the UK to report small area statistics, and are defined to encompass areas with approximately 1500 residents; the mean area of the 2946 rural

LSOAs in England is 30.8 km². The 12,697 estimation observations were 52.40% female; mean age 47.59 years old; 72.73% married or living with a partner; 59.65% in employment and 21.14% retired (see Supplementary Table 1 for full details). The observations of BHPS respondents from rural England that could not be included in the sample ($n = 7903$) had a greater proportion of females (56.99%), and less people in employment (54.04%) and more in retirement (26.02%) than the estimation sample, but were only slightly older, with a mean age of 48.74 years old, and there were no other differences of note. Thus caution is required in the interpretation of the current results as representative of the rural population.

Importantly, of the total sample, 214 individuals relocated from one rural area to another during their time in the panel. This meant that these individuals had potential change in both (a) the land cover make-up of the areas in which they resided and (b) their mental health. Due to the longitudinal nature of the panel this sub-sample was associated with 1867 observations ($M = 8.7$ observations per person) which could thus be examined for within-individual differences. Descriptive statistics for this subset of the estimation sample observations were examined to determine whether they were different in important ways from the full set of estimation sample observations. These observations were 51.74% female; mean age 42.17 years old; 79.33% married or living with a partner; 72.09% in employment and 9.80% retired (see Supplementary Table A for full details). Thus, the subsample of the estimation sample observations from which within-individual land cover effects were estimated were, on average, younger, more likely to be in employment, and less likely to be retired, than the full set of estimation sample observations. Not all of the 214 individuals who relocated between rural areas experienced change on all land cover classes; much relocation was both from, and to, areas with zero coastal land cover, for example. The sample size from which within-individual effects of each specific land cover type are estimated, are presented with results.

2.2. Measures

2.2.1. Land cover

The current research used the ten aggregate land cover classes in the Land Cover Map 2007 (LCM2007; Morton et al., 2011): Broadleaf woodland; Coniferous woodland; Arable; Improved grassland; Semi-natural grassland; Mountain, heath and bog; Saltwater; Freshwater; Coastal; and Built-up areas (including gardens). Although LCM2007 also provides a finer grained 23 category taxonomy of land cover types, it is less reliable, and the ten categories were deemed sufficient for present purposes. LCM2007 maps land cover types across the UK using 25 m grid cells. The map is created by classifying summer-winter composite satellite images of polygons. A spatial framework based on generalised digital cartography, and also on agricultural census data, is used to ensure that the map is constructed from polygons which represent meaningful real-world land parcels, such as fields and plots of woodland.

In the current study, GIS areal interpolation (Flowerdew, Green, & Kehris, 1991) was used to assign the 25 m grid cells of LCM2007 to LSOAs. The percentage of each LSOA comprised of each of the ten land cover classes was then calculated. LCM2007 includes an Unclassified category which was disregarded from the percentage base; when assigned to LSOAs, only 169 of the 2946 rural LSOAs included some unclassified land, which accounted for $M = .0565\%$ land cover in the LSOAs affected ($SD = .1246\%$). Finally, these LSOA land cover percentage measures were linked to the BHPS sample observations based on residential area LSOA.

Descriptive data on the mean proportions of the ten land cover classes for both all rural LSOAs in England, and for the estimation sample observations, are presented in Table 1. In addition, a

Table 1

Land cover by LCM2007 aggregate class and total generic natural space (mean percentage) of all English LSOAs, and of estimation sample observations.

LCM2007 aggregate class	Rural LSOAs (<i>n</i> = 2,946), mean % land cover (<i>SD</i>)	Estimation sample observations (<i>n</i> = 12,697), mean % land cover (<i>SD</i>)
Broadleaved woodland	8.21 (7.66)	8.97 (9.36)
Coniferous woodland	1.77 (3.88)	1.41 (2.61)
Arable	46.92 (23.56)	43.72 (23.91)
Improved grassland	29.63 (16.21)	32.10 (17.49)
Semi-natural grassland	6.20 (6.27)	6.58 (5.00)
Mountain, heath and bog	2.40 (7.75)	2.43 (6.69)
Saltwater	.02 (.14)	.03 (.14)
Freshwater	.59 (1.86)	.63 (1.33)
Coastal	.43 (2.84)	.31 (1.75)
Total Generic 'natural space' ^a	96.16 (5.76)	96.19 (5.31)
Built-up areas and gardens	3.84 (5.35)	3.81 (5.31)

^a Aggregates all the LCM2007 aggregate classes except: Built-up areas (including gardens).

measure of generic natural space was derived by collapsing all the land cover categories except Built-up land; we thus define 'natural space' to include Freshwater, Saltwater, Coastal etc. rather than limiting the definition to a narrower subset of land cover categories comprising green vegetation, which might more commonly be grouped together as generic green space. There are no differences of note between the mean proportions of the land cover classes, or the generic measure, amongst the estimation sample, and the sample of all rural LSOAs, and in this respect the residential areas of the panel respondents living in rural areas are broadly typical of such residential areas nationwide.

2.2.2. Mental health

Mental health was measured with the short form, 12 item General Health Questionnaire (GHQ-12), a widely used and reliable self-assessment screening tool to aid clinical diagnosis of mood disorders such as anxiety and depression (Goldberg et al., 1997). It is also commonly used as a measure of positive mental health in epidemiological research with population samples and is robust to re-test effects (Pevalin, 2000). Respondents are asked to consider how the "past few weeks" compared to "usual" in terms of 6 positive and 6 negative states, e.g. coping. For each item there are four ordinal response options, scored 0, 1, 2 or 3. Item scores were summed to give interval scale scores in the range 0–36; the scale was reversed in the current study so that higher GHQ scores as we report them (i.e. inverse-GHQ) represent better mental health. The estimation sample observations (*n* = 12,697) had a mean GHQ score of 25.10 (*SD* = 5.19), showing on the average, good mental health.

As a robustness check, mental health was also operationalised from GHQ-12 item responses using a second standard scoring method (Goldberg & Williams, 1988) to derive an indicator outcome variable. The two item responses considered to reflect low risk of poor mental health were scored 0, and the two considered to reflect a higher risk of poor mental health were scored 1. These item scores were summed to give totals in the range 0, very good mental health, to 12, very poor mental health (estimation sample mean = 1.702; *SD* = 2.844), and a categorical measure of poor mental health or 'psychiatric caseness' was defined as a score ≥ 3. This 'caseness' criterion is met by 2951 (23.24%) of the estimation sample observations.

2.2.3. Control variables

Regression models included control variables at both area and individual levels, for factors known to be related to GHQ (Dolan, Peasgood, & White, 2008). LSOA level controls were taken from the English Indices of Deprivation (ID 2010 release; Department of Communities & Local Government, 2008), and linked to the sample observations based on residential area LSOA: income deprivation (derived from social benefit data); employment deprivation (derived from unemployment data); education deprivation (derived from school performance data, participation rates in higher education and working age adult qualifications data); and crime (derived from crime records). Individual level control variables included factors known to be related to GHQ, and factors that may change upon relocation and so potentially confound the effects of land cover: Age category (25 and under; 26–35; 36–45; 46–55 – reference category; 56–65; 66–75; over 75); Diploma/degree level qualified; Married/living with a partner; Living with an own child aged under 16; Household income operationalised as the log of net annual household income in the preceding 12 months adjusted for household composition using the Before Housing Costs equivalence scale, and indexed to January 2010 prices (Jenkins, 2010); Work-limiting health; Labour market status (employed/self-employed – reference category; unemployed; retired; in education/training); Residential accommodation (detached house – reference category; semi-detached house; terraced house; flat; other); Accommodation size (less than 1 room/person; 1 – less than 3 rooms/person – reference; 3 or more rooms/person); Commuting time (non-commuter – reference category; 15 min and less; 16–30 min; 31–50 min; over 50 min); and Fixed time effects (indicator variables for BHPS data collection waves).

2.3. Analytic approach

Regression analyses were conducted using the xt suite of functions in Stata 12 (StataCorp, College Station, TX). Random Effects (RE) regression was used to estimate the effects of land cover on GHQ interval scale scores. Separate estimates were derived based on differences between the mean scores of different individuals (between-individual effects), and based on scores for the same individuals at different points in time where individuals relocated from one rural area to another and so had within-individual variation on the land cover variables of interest (within-individuals effects). These separate effects estimates were derived by inclusion in the model of individual mean values for all independent variables (Schunck, 2013; Rabe-Hesketh & Skrondal, 2008). The model for GHQ can be expressed by

$$GHQ_{ij} = \beta_0 + \beta\bar{X}_{1i} + \beta\bar{X}_{2i} + \beta(X_{1ij} - \bar{X}_{1i}) + \beta(X_{2ij} - \bar{X}_{2i}) + (\mu_{0j} + \varepsilon_{ij})$$

where β_0 denotes the model intercept; $i = 1, 2, \dots, n$ denotes the individual respondent; $j = 1, 2, \dots, n$ denotes the observation for the individual; GHQ_{ij} denotes the GHQ response for observation j for individual i ; $\bar{X}_{1i}$ denotes the mean values of the set of land cover exposure measures for all observations for individual i ; $\bar{X}_{2i}$ denotes the mean values of the set of individual and area level control measures for all observations for individual i ; X_{1ij} denotes the set of land cover exposure measures for observation j for individual i ; and X_{2ij} denotes the set of individual and area level control measures for observation j for individual i . Then, the between-individual effects of land cover exposure are given by $\beta\bar{X}_{1i}$, and the within-individuals effects of land cover exposure are given by $\beta(X_{1ij} - \bar{X}_{1i})$. The significance of differences between the estimated within-individual and between-individual effects was tested by substituting in the model,

non-individual mean centred values, in place of individual mean centred values:

$$GHQ_{ij} = \beta_0 + \beta\bar{X}_{1i} + \beta\bar{X}_{2i} + \beta X_{1ij} + \beta X_{2ij} + (\mu_{0j} + e_{ij})$$

so that the difference in the within-individual and between-individual effects of land cover exposure are given by $\beta\bar{X}$ (Rabe-Hesketh & Skrondal, 2008; Schmpf & Kaufman, 2012).

The estimated within-individual effects of land cover control for personality, upbringing and other stable factors which may influence both an individual's mental health and where he or she chooses to live. Importantly, in the multiple land cover model, we follow Mourato et al. (2010) in omitting the Built-up land cover category. The regression coefficients therefore represent the scale point change in GHQ for a percentage point increase in a given land cover type whilst decreasing by a percentage point the share of Built-up land cover, when other independent variables (including other land cover types) are held constant.

3. Results

3.1. Generic natural space

The initial exploratory analysis examined natural space, which is broadly comparable to a generic green space measure. The between-individuals estimate of this measure had no significant relationship with GHQ; a 1% natural space increase was associated with a .01 scale point increase in GHQ ($SE = .02$; $p = .619$). However, a significant association was found from the within-individuals estimate, where a 1% natural space increase was associated with a .092 scale point increase in GHQ ($SE = .04$; $p = .021$). This result was robust to the caseness measure, where no significant between-individuals effect was observed, but the estimate from within-individual differences showed a 1% increase in natural space was associated with decreased odds of achieving psychiatric caseness ($OR = .954$; $SE = .02$; $p = .025$). Thus, despite relatively high levels of green and other natural space in all rural areas and the lack of any relationship between the neighbourhood natural space density of different individuals and their mental health, increases in rural natural space experienced by individuals were associated with significant benefits to mental health. Subsequent analyses

investigated whether this relationship was driven by all non-Built land, or whether the different types of rural land cover were differentially related to within-individual change in mental health.

3.2. Land cover types

Estimates for the main model, which specified simultaneous entry of the land cover classes, are summarised in Table 2 (see Supplementary Information Table 2 for full details). The between-individual estimates showed no significant associations between any land cover type and mental health. However, the within-individual estimates showed significant ($p < .05$) positive associations with Arable, Improved grassland, Semi-natural grassland, Mountain, heath and bog, and with Coastal land cover. The relative benefits of these types of land cover over Built-up land, adjusted for their prevalence in the wider estimation sample, can be compared by examining the scale point increases in the 0–36 GHQ measure associated with 1 *SD* increases in these land covers: Arable, 1.988; Improved grassland, 2.363; Semi-natural grassland, .763; Mountain, heath and bog, 1.116; Coastal, 1.463 (Table 2). Furthermore, a significant negative association was observed with Saltwater, with a 1 *SD* increase associated with a decrease in GHQ of .346 scale points. The relationships estimated from differences between individuals regarding GHQ and the Improved grassland, Mountain, heath and bog, Coastal and Saltwater land cover classes, were significantly different from those estimated from differences within-individuals at different times.

3.3. GHQ robustness check

A robustness check for the main model examined relationships between the land cover types and the GHQ caseness binary outcome (Supplementary Table 3). Within-individual estimates showed significantly lower odds ($p < .05$, compared to the Built up category) of reporting psychiatric caseness to be associated with Broadleaf woodland, Improved grassland, Semi-natural grassland and Coastal land cover, and significantly higher odds associated with Saltwater, whilst the lower odds associated with Mountain, heath and bog were marginally significant ($p = .055$). The main model estimates from within-individual differences of significant

Table 2
Estimation sample descriptive data, and random effects regression estimates of GHQ (0–36) in rural areas against LCM2007 aggregate land cover classes^a; number of observations = 12,697; number of individuals = 2020.

	Estimation sample <i>SD</i>	Between-individual effects (individual mean variables)				Within-individual effects (individual mean-centred variables)				Significance of difference in within- and between-individual effects	
		Coeff.	SE	<i>p</i>	Effect of 1 <i>SD</i> increase	Within-individual difference no. of individuals (no. of obs.)	Coeff.	SE	<i>p</i>	Effect of 1 <i>SD</i> increase	<i>p</i>
Broadleaved woodland	9.36	.0071	.0271	.794	.063	214 (1867)	.0803	.0616	.192	.721	.282
Coniferous woodland	2.61	-.0248	.0465	.594	-.065	208 (1817)	.0766	.0712	.282	.200	.234
Arable	23.91	.0115	.0211	.587	.274	214 (1867)	.0831	.0369	.024	1.988	.097
Improved grassland	17.49	.0052	.0219	.811	.092	214 (1867)	.1351	.0396	.001	2.363	.005
Semi-natural grassland	5.00	.0321	.0334	.337	.161	214 (1867)	.1524	.0615	.013	.763	.097
Mountain, heath and bog	6.69	-.0041	.0266	.877	-.028	193 (1667)	.1667	.0738	.024	1.116	.027
Saltwater	.14	.3398	.7027	.629	.049	37 (351)	-2.4219	1.1305	.032	-.346	.035
Freshwater	1.33	-.0289	.0693	.677	-.038	195 (1712)	.4700	.3545	.185	.625	.173
Coastal	1.75	.0298	.0818	.716	.052	46 (412)	.8380	.3318	.012	1.463	.019

^a Coefficients in columns 2 and 7 represent fitted change in GHQ score for a 1% increase in land cover classes (within and between individuals respectively) and a corresponding 1% decrease in urban land cover, and columns 5 and 10 represent fitted change in GHQ score for a 1 *SD* (see column 1) increase in land cover classes; model includes controls for: Government office region; Area income deprivation; Area employment deprivation; Area education deprivation; Area crime deprivation; Age; Education; Marital status; Parenting status; Household income; Work-limiting health; Labour market status; Residential accommodation; Commuting; Fixed time effects.

benefits to mental health associated with the grassland land cover classes and Coastal land cover, were therefore found to be robust to the caseness measure, but the estimated benefit of Mountain, heath and bog was statistically weaker. The effect of Saltwater on mental health observed in the main model was also robust to the caseness measure. However, the significance of beneficial relationships with Arable and Broadleaf woodland differed when the interval scale and binary GHQ outcomes were regressed against them, with Arable significantly associated only with the former, and Broadleaf woodland significantly associated only with the latter. The lack of significant associations from between-individual estimates in the main model was replicated in the robustness check.

4. Discussion

4.1. Summary of results

GIS was used to link high spatial resolution land cover data in ten classes (e.g. woodlands, arable) to rural LSOAs in England to enable a novel analysis of relationships with an indicator of mental health. Relationships were estimated for a longitudinal sample of residents of rural LSOAs, some of whom moved from one rural area to another and so had different land cover exposure at different times. By specifying individual mean values on predictor variables in conjunction with individual mean-centred values, separate effects estimates for both natural space and a set of rural land cover types were derived from between-individual differences and within-individual differences at different times.

Natural space was positively related to good mental health when estimated from within-individual differences, supporting the hypothesis that even in rural residential areas levels of green and other natural space can be associated with this wellbeing measure. The main model suggested different relationships between different types of land cover and mental health. Replicating earlier findings which included both urban and rural areas (White et al., 2013b), individuals who moved home during their time in the panel (i.e. within-individual effects) showed better mental health in years when they were living in areas with more coastal land such as beaches. In contrast, and unreported by previous work, Saltwater land cover including mudflats exposed when tidal rivers are at low tide, had a negative within-individuals effect, although it must be noted that the samples from which within individual effects of Coastal and Saltwater were estimated were very small (only 412 observations from 46 individuals, and 351 observations from 37 individuals, respectively). The 'open' landscapes found where land cover was Improved or Semi-natural grassland, or Mountain Heath and Bog, also showed positive within-individual effects. As these land cover types will potentially afford wide views to relatively distant horizons, this is consistent with the Prospect-Refuge theory of landscape preference (Appleton, 1984). Arable also showed positive within-individual effects, though this was not robust to the caseness measure.

Confidence in these results comes from the estimation of associations with socio-demographic covariates from both between-individual and within-individual variation which replicate previous findings (Supplementary Information Table 2; Dolan et al., 2008; White et al., 2013a). Examples include the significant positive associations between mental health and being employed rather than unemployed, and the significant negative associations with having a work-limiting health condition rather than not.

4.2. Links to other research

In general, the green land cover types associated with mental health improvements within individuals were consistent with the

results of the Swedish research using the Scania Green Score. Whilst the detail of the SGS derivation is complex and involves other inclusion criteria (Skärbäck et al., 2012), the SGS operationalised desirable qualities in natural environments in terms of categories from the Swedish Corine Landcover Project (SCLP; Ahlcróna, Olsson, & Rosengren, 2001). Ten categories of SCLP land parcel which were considered to offer desirable recreational qualities will be listed, and their likely equivalent LCM2007 aggregate class will be stated where this is not obvious; it will be noted that the LCM2007 equivalents of these categories showed evidence of positive association with good mental health: (1) 'Non-urban parks', (classified as Improved grassland by LCM2007); (2) 'Pastures', (classified as Improved grassland by LCM2007); (3) 'Broad-leaved forest'; (4) 'Mixed forest'; (5) 'Natural grassland'; (6) 'Moors and heathland'; (7) 'Thickets'; (8) 'Open spaces with little or no vegetation' (which includes beaches, classified as Coastal, and bare rock, classified as Mountain, heath and bog by LCM2007); (9) 'Wetlands' (which include marshes, classified as Semi-natural grassland, and salt marshes, classified as Coastal by LCM2007); and (10) 'Inland water'. With the exception of the LCM2007 Freshwater land cover category, which is scarce in England, the LCM2007 equivalents of these SCLP categories showed evidence of positive association with good mental health, although significant association with the LCM2007 Broadleaf woodland category was only with the caseness measure. The SCLP classes which are specifically excluded from the SGS are 'Artificial surfaces' (classified as Built-up by LCM2007); 'Coniferous forest'; 'Arable', though there are exceptions to this where ancient farmland is environmentally protected; 'Permanent crops' (classified as Arable by LCM2007); and 'Marine' waters. Again, there is some consistency here with our findings, since the LCM2007 category Coniferous woodland had no significant association, and Saltwater had a negative association.

Our within-individual findings are also consistent with the research literature on landscape preferences. Preferences are not trivial, since, from an evolutionary perspective, organisms adapt to enable them to survive and reproduce in landscape types where they are able to exploit resources. Thus, it is argued that despite variation in people's expressed preferences according to factors such as prior experience, culture and demographics, landscape preferences indicate settings in which people are more likely to function effectively (Kaplan & Kaplan, 1989). It is for this reason that preferences drive decisions we make in life, such as, for instance, where we choose to live. Some studies have demonstrated that people tend to prefer parkland and savannah-like landscapes (Falk & Balling, 2010), and similar types of open, lightly wooded, green landscape have been idealised throughout human civilisation (Ward Thompson, 2010). When asked to categorise images of environments, some of the most consistent differentiation is between 'wild' and more ordered and managed landscapes (Hartig & Evans, 1993), suggesting that degree of human influence is relevant in people's responses. For instance, van den Berg and Koole (2006) and de Groot and van den Born (2003) reported that wild landscapes were generally rated more beautiful than managed landscapes. Similarly, a Norwegian study (Kaltenborn & Bjerke, 2002) of preferences in a rural community found images with features such as mountains, forests and lakes, were rated as most attractive, and modern agriculture received the lowest ratings.

Preferences for different agricultural landscapes have also been compared. Examining Swiss alpine landscapes, Lindemann-Matthies, Briegela, Schupbach, and Junge (2010) found low-intensively managed, species-rich grassland were rated highest, and mere production landscapes with high-intensity grassland and arable land were rated lowest. In addition, a Northern Ireland study found images of 'traditional' farming landscapes, which included grassland categories, were preferred over 'intensive' agriculture, which included arable categories (Howley, Donoghue, & Hynes,

2012). In sum, this research tradition indicates the unpopularity of more managed and specifically arable landscapes, in contrast to the popularity of land cover such as grassland, which may have savannah-like characteristics. These preference trends are broadly consistent with associations we observed between land cover and positive mental health amongst those relocating between rural areas. Whilst Arable land cover, which is excluded from the SGS, was associated with good mental health compared to Built-up land cover (on the interval scale measure only), the within-individual beneficial effects associated with an increase in Arable were substantially below those associated with the grassland categories. For instance, using the current data, a planned comparison test which specified Arable rather than Built-up as the reference showed that a 1% increase in Improved grassland over Arable was associated with an increase of .0519 GHQ scale points ($SE = .0178$; $p = .004$).

4.3. Accounting for the observed contrast in the between and within individual effects

Given previous findings on landscape preferences, and the within-individuals estimates, we are unsure why land cover was not related to differences in mental health between individuals. One possibility is that differences in the two samples contributed to these estimates. It was noted above that those relocating between rural areas are younger and more likely to be in employment, and possibly this difference is important in this respect. They may also differ in factors which are not controlled in the models. A potentially relevant factor is residential area attachment, which is related to mental health (Pevalin, 2003; O'Brien, Hassinger, & Dershem, 1994), and may also be related to land cover. It may be, for instance, that neighbourhood attachment was lower amongst those who relocated and, precisely because of this, it was these individuals who sought to move to an area they would like more and which would make them happier. This is speculation at this point however, and further work is needed to understand the motivations behind why people do or do not move to different areas, and to specific environment types.

Another possibility is that psychological adaptation (Diener, Lucas, & Scollon, 2006) occurs after one has lived in an area with positive land cover features (e.g. broadleaf woodlands and coastal) for a number of years and thus the between individual differences become smaller the longer people have lived in the same area. Adaptation was not observed within a 3-year period following increases in urban green space amongst intra-urban movers, but adaptation was observed following reductions in green space (Alcock et al., 2014), and it might be that adaptation to land cover change does occur in rural areas. Again, it would be interesting to see how long our movers would need to live in their new areas before adaptation may occur.

Finally, a third reason why between-individual effects were not found may be, in part, statistical. Specifically, individual heterogeneity was not accounted for in the between participant estimates and it may be that this is an important omission. For instance, it may be that by not controlling for things such as upbringing and personality, there is simply too much 'noise' in the data for any effects of land cover type to emerge. This would be particularly important if there was indeed a difference in upbringing or other non-observed factors which were influencing both where individuals lived and their mental health. For example, if certain types of people with generally better mental health lived in areas with less woodland, this might, at the aggregate level, mask a statistically significant beneficial effect of living near woodlands for those with poorer mental health. Thus one may not see a difference in land cover because there are important non-observed differences in people who reside in different areas.

4.4. Implications and limitations

Despite the lack of between individual effects, the evidence from the within-individual estimates that different types of green land cover are differentially related to mental health in rural areas has potential implications for the framing of policy designed to maximise health benefits from natural environments. Current planning and health promotion policies emphasise increasing access to generic green space in urban areas; see, for example, Department of Health (2011) in the UK, and the American Public Health Association draft policy statement on Nature, Health and Wellness in the US. The policy focus to date on generic green space in urban areas doubtless follows, in part, from the research evidence pointing to the importance of exactly this. It would also be fair to point out that it is in the urban parts of economically developed nations that large proportions of the population, as well as the burden of disease amongst them, are concentrated. Furthermore, green space in urban areas is often a threatened or dwindling resource in many countries, creating a pressing need for evidence-based urban planning policy which can work in conjunction with health policy (Lee & Maheswaran, 2010).

However, policies which address generic green space may, by definition, neglect important contributions from natural environments to health in rural areas. Whilst generic green space covers a far greater proportion of rural LSOAs than urban LSOAs in England, (mean 96.16% versus 27.82%), the nature of 'green space' differs markedly. Urban green space in the UK is typically characterised by parks with grassland and trees, gardens (which may be included as a component in generic green space), and small pockets of undeveloped natural habitats, allotments, roadside verges and river corridors. Arable is a category of green landscape which dominates many rural residential areas but is a relatively minor constituent of urban green space. In other words, policy references to urban green space are often to improved grassland, whereas references to rural green space are more likely to be to arable land. A more nuanced policy approach might acknowledge the importance of the nature of green space in rural areas.

Despite the strengths of our analysis, we recognise a number of limitations. The analysis considers only main effects. Cumulative or synergistic effects of different land cover types were beyond the scope of this study. Further, LCM2007 measures of land cover collected between 2005 and 2008 (Morton et al., 2011) are linked to individual respondent data collected between 1991 and 2008. Longitudinal change in land cover is therefore not accounted for. In addition, the level of geographical resolution used in the production of LCM2007 means that the distribution maps have a degree of simplification, since only UK land parcels over 5000 m² are classified and mapped, with smaller parcels 'dissolved' into the surrounding landscape. Moreover, this study is premised on the assumption that residential area land cover is a valid proxy measure for individual exposure. No measures were available from the BHPS data of factors relevant to this assumption, such as how much time respondents spent out of doors. Whilst LSOA is a typical unit of analysis used in this type of research (Mitchell & Popham, 2008), there are also drawbacks related to the use of LSOA as the residential unit, since rural LSOAs vary greatly in size; amongst the estimation sample observations, these areas had $M = 29.5$ km² and $SD = 23.4$ km². Thus, whilst the average residential area within which land cover is operationalised is an area with a radius of approximately 3 km, a residential area boundary used in previous research on green space and health (Maas et al., 2006), some individuals are linked to land cover data from substantially larger areas. Furthermore, areas are not equidistant buffers around residences, so that the land cover linked to one individual may be mainly to the north of the home, whilst that linked to another may be mainly to the south, etc. This lack of consistent definition of the land cover exposure variables

arises from the use of an LSOA level dataset which was prepared for another purpose, and any bias resulting from this is unknown. Future research might use a much more precise exposure proxy based on the derivation of buffers around residential postcodes.

Although one strength of the analysis is that the within-individual estimates control for factors such as personality, a weakness is that the sample of individuals who contribute to these estimates was relatively small, and it would be inappropriate to assume them to be representative of the population. Moreover, even in the case of a representative sample, it is important to note that within-individual estimates would not permit inference to a sampled population. The successor survey to the BHPS, the Understanding Society panel survey, has a much larger sample, and over time the subsample of individuals who engage in the relatively rare event of relocation from one rural area to another will be larger in this panel than in the BHPS, allowing estimates to be based on a greater number of observations as well as allowing increased understanding through sample stratification on demographic factors such as age and socio-economic status. A further limitation is that whilst the within-individual estimates control for time-invariant individual effects, we were unable to control for all potentially relevant time varying factors, especially at the area level, and so these effects cannot be assumed to be causal effects.

5. Conclusion

This research focused on land cover in rural residential areas, and so provided a context where green space is more varied than in urban areas which are usually the setting for research in this field. Land cover data on residential areas was linked through GIS to repeated measures of individuals' mental health status. Natural space in rural areas was found to be positively related to good mental health when estimated from within-individual differences. The results from re-locating individuals also offer some evidence that different types of green and other natural space offer different degrees of benefit to human wellbeing. However, no evidence in support of this hypothesis emerged from comparing land cover exposure between individuals. These results, and the inherent limitations in the study design, underline the need for further research to better inform land development policies.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.landurbplan.2015.05.008>

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The effect of climate change on indicators of fire danger in the UK

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Keywords: climate change, wildfire, UK, UKCP18Supplementary material for this article is available [online](#)**Abstract**

The UK is vulnerable to wildfire, and vulnerability is likely to increase due to climate change. Whilst the risk is small compared with many other countries, recent fires have raised awareness and highlighted the potential for environmental damage and loss of property and key infrastructure. Most UK wildfires are a result of inadvertent or deliberate human action, but the environmental conditions depend on antecedent and current weather. This paper presents projections of the effects of climate change on UK wildfire danger, using a version of an operational fire danger model, UKCP18 climate projections representing low and high emissions, and several indicators of fire danger. Fire danger will increase across the whole of the UK, but the extent and variability in change varies with indicator. The absolute danger now and into the future is greatest in the south and east (the average number of danger days increases 3–4 times by the 2080s), but danger increases further north from a lower base. The variation in change across the UK for indicators based on absolute thresholds is determined by how often those thresholds are exceeded now, whilst the (lesser) variability in percentile-based indicators reflects variability in the projected change in climate. Half of the increase in danger is due to increased temperature, and most of the rest is due to projected reductions in relative humidity. Uncertainty in the magnitude of the change is due to uncertainty in changes in temperature, relative humidity, and rainfall, and there is a large difference between two of the UKCP18 climate model ensembles. Reducing emissions to levels consistent with achieving international climate policy targets significantly reduces, but does not eliminate, the increase in fire danger. The results imply that greater attention needs to be given to wildfire danger in both emergency and spatial planning, and in the development of guidelines for activities that may trigger fires. They suggest the need for the development of a fire danger system more tailored to UK conditions, and the combination of fire danger modelling with projections of sources of ignition to better estimate the change in wildfire risk.

1. Introduction

In recent years there have been widespread and extensive wildfires in many countries and regions, notably Australia, California, Brazil, Greece and Siberia. Whilst the global number of wildfires has decreased over the last few decades (Doerr and Santin 2016), the risks posed by wildfire to the environment, health and the economy are gaining increased attention, and climate change has the potential to increase substantially the risk of wildfires. The apparent contradiction between observed global trends, recent experience and concerns over the future arises

because wildfire is a semi-natural hazard: whilst the conditions necessary for a fire to become established are largely environmentally-determined, the initial spark is often a result of deliberate or inadvertent human action.

The UK is vulnerable to wildfires, and awareness is increasing (Glaves *et al* 2020). Following the Swinley Forest fire in May 2011 that threatened critical infrastructure in a rural-urban margin in south east England, wildfire risk was added to the UK's National Risk Register in 2013 (Gazzard *et al* 2016): however, fewer than half of the 42 current Community Risk Registers in England and Wales

explicitly mention fire. The Saddleworth Moor fire in May 2018 threatened homes and led to evacuations, and an extensive wildfire in Moray, Scotland, in April 2019 threatened an onshore wind farm. A wildfire in the UK is rather broadly defined by the Scottish Government as ‘any uncontrolled vegetation fire which requires a decision, or action, regarding suppression’ (Scottish Government 2013), and most incidents are very small: 99% are under 1 ha, and almost half are under 5 m². Between 2009/10 and 2016/17 there were over 258 thousand wildfires in England, burning nearly 37 thousand hectares (Forestry Commission 2019). The European Forest Fire Information Service (EFFIS) counts fires greater than 30 ha: an average of just over 10 000 ha was burnt in such fires across the UK between 2011 and 2020, with 28 754 ha burnt in 2019. Most of the land area burnt by wildfires across the UK is arable, grassland, or mountain and heath open habitats (de Jong *et al* 2016, Gazzard *et al* 2016, Forestry Commission 2019). Between 2009/10 and 2016/17 woodland and forest fires accounted for less than 5% of the land area burnt in England (Forestry Commission 2019). The vast majority of wildfires in the UK therefore occur in areas with low shrub vegetation (i.e. lowland and upland heath), and the dramatic scenes of fire leaping from crown to crown across a forest are very rare: the Swinley Forest fire was a notably significant exception.

There is little information on the costs and consequences of wildfire in the UK, although Gazzard *et al* (2016) estimated that suppression costs up to £55million yr⁻¹. In principle wildfire in the UK can threaten life, and damage infrastructure and property, but the actual losses have been very low. The greatest impacts of wildfire in the UK to date have been on the environment: fires damage wildlife habitats and landscape quality, release carbon into the atmosphere and trigger soil erosion. The UK does not have dedicated resources to deal with wildfires, but the England and Wales, and Scottish, Wildfire Forums are voluntary strategic bodies established to coordinate interested stakeholders including local councils and the fire and rescue services.

Wildfire ‘risk’—expressed in terms of area burnt or some other loss—is a function of two factors: does a fire start, and how far does it go? The chance of a wildfire starting depends on whether there is a source of ignition and whether the conditions are favourable for the fire to develop. In the UK, virtually all wildfires are started by human action (Glaves *et al* 2020), either inadvertent (a discarded barbecue, or a controlled burn that gets out of control, for example) or deliberate (malicious, although prosecutions for arson are very rare): fires could be started by lightning, but this is very rare. Once initiated, the extent of an uncontrolled wildfire depends on when it runs out of fuel, when it rains, or - most often - when it is suppressed by fire and rescue services. Fire is widely used as a traditional habitat management tool in moorland

parts of the UK (Davies *et al* 2016); most are set in winter or early spring.

Estimating wildfire risk therefore involves understanding and estimating human behaviour as well as the factors which influence susceptibility to fire. In practice, operational wildfire warning systems are based on forecasting fire ‘danger’: the chance of a fire developing once there is a source of ignition. This chance depends on the availability of dry combustible material, which itself depends on the type of vegetation and current and recent weather. Fire danger for a given vegetation type can therefore be estimated from meteorological data.

Surprisingly few studies globally have assessed the effects of future climate change on wildfire danger. Virtually all have used indicators of fire danger (Flannigan *et al* 2013, Wang *et al* 2015, Khabarov *et al* 2016, Bowman *et al* 2017, de Rigo *et al* 2017, Wang *et al* 2017a, Kerr *et al* 2018, Dowdy *et al* 2019, Fonseca *et al* 2019), and all have demonstrated increased danger in the future. Albertson *et al*’s (2010) study in the Peak District of the UK used an empirical statistical model linking actual fire occurrence to temperature, precipitation and date (as a surrogate for visitor numbers), thus combining empirically the chance of ignition with the chance of a fire developing.

This paper presents an analysis of the effects of climate change on a series of indicators of wildfire danger in the UK. These indicators are currently used operationally to assess fire danger. The paper (a) evaluates change in fire danger across the UK with high future emissions of greenhouse gases, (b) evaluates the effect of reducing emissions on fire danger, and (c) identifies the main climatic drivers of change in fire danger in the UK. The paper forms part of a larger multi-sectoral evaluation of changing climate risks in the UK (Arnell *et al* 2021).

2. Indicators, data and scenarios

2.1. Overall approach

Wildfire danger in the UK is characterised by a series of indicators currently used in operational fire danger forecasting by the UK Meteorological Office (2003), using 12 × 12 km gridded observed daily climate data for 1981–2010 and the 2018 UK Climate Projections (UKCP18) applied using the delta method. Results are aggregated to the regional level, weighted by land cover.

2.2. Characterising the potential severity of wildfires

A Fire Severity Index (FSI) service (based on an assessment of how severe a fire could become if one were to break out) for the UK is delivered by the Met Office using the Met Office FSI (MOFSI) model. The service, delivered under contract to Natural England and Natural Resources Wales, is used to inform the

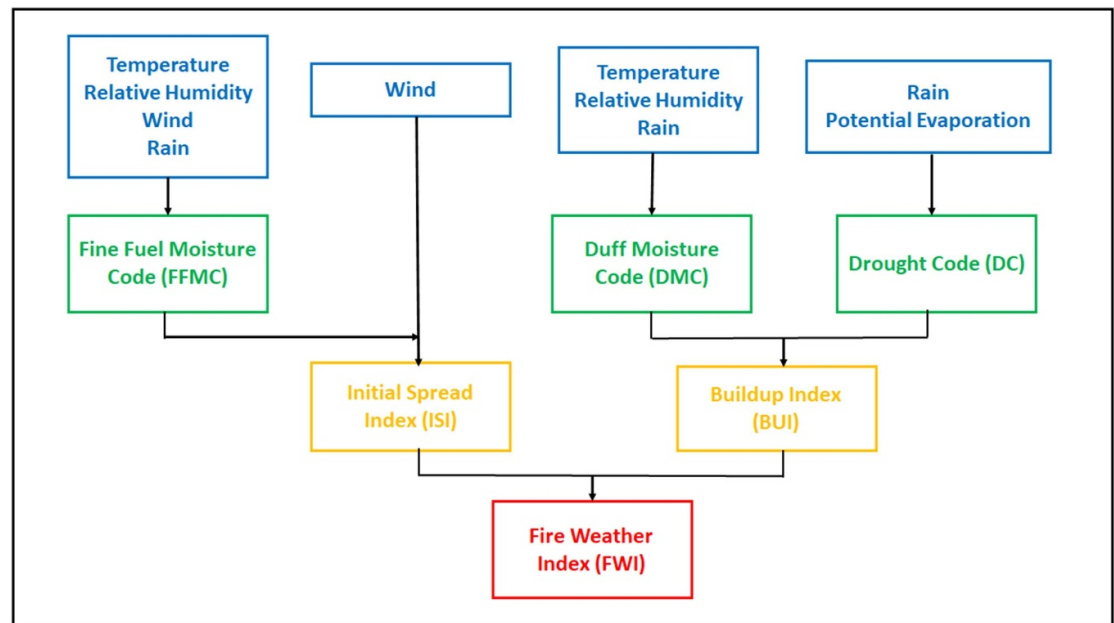


Figure 1. The Canadian Forest Fire Weather Index System, as used in the Met Office Fire Severity System and implemented in this paper.

triggering of public restrictions to open access land, as specified under the Countryside and Rights of Way Act (2000). The act allows open public access to specific areas of land, but permits landowners to restrict access for a number of reasons, including fire danger. MOFSI categorises danger into four levels, using consistent definitions across England and Wales. Land owners can restrict access when “exceptional” danger is forecast. Maps showing current and forecast FSI levels are published daily by the Met Office. MOFSI ‘exceptional’ conditions are currently rare in the UK, and recent large wildfires have occurred with warnings at ‘very high’ or even ‘high’ levels (for example the Saddleworth fire in 2018).

MOFSI is based on the Canadian Forest FWI System (de Groot 1987, Van Wagner 1987, Wang *et al* 2017b), summarised in figure 1. Fire danger is calculated from temperature, relative humidity, wind speed, rainfall and potential evaporation, and the system consists of a series of indices. The Fine Fuel Moisture Code (FFMC) index represents the availability of dry ‘tinder’ material in and around the vegetation susceptible to ignition. The Duff Moisture Code (DMC) characterises the moisture content in partially decomposed loose material at the surface, whilst the Drought Code (DC) characterises the moisture content in the soil and vegetation. These vary at different time scales: the FFMC is sensitive to day to day variations in weather, whilst the DC reflects water deficits over weeks and months. The Initial Spread Index (ISI) combines the FFMC with windspeed to characterise the chance of a spark becoming a fire. The Buildup Index (BUI) integrates the DMC and the DC to characterise the potential for the fire to develop.

The overall Fire Weather Index (FWI) combines the ISI and BUI. Table 1 summarises the threshold values for the FWI used in MOFSI to define ‘very high’ and ‘exceptional’ danger. The model and algorithms were developed for Canadian conditions, mostly based on wildfires in jack pine forest. Although wildfire processes and fuel sources in the UK are very different, there is evidence that MOFSI and its component parts can successfully predict extreme fire weather conditions in the UK (Glaves *et al* 2020). Davies and Legg (2016) and de Jong *et al* (2016) both concluded that high values of FFMC, coupled with high ISI (Davies and Legg 2016), were better at predicting fire occurrence for some fuel sources than the summary FWI.

The Canadian system is also used in the European Forest Fire Information System (EFFIS: San-Miguel-Ayán *et al* 2012) and in other countries including New Zealand (Pearce and Clifford 2008). EFFIS issues fire danger forecasts in the same way to MOFSI, using a similar threshold (50) for ‘exceptional’ conditions but with a substantially higher FWI threshold (38) for ‘very high’ danger.

The Natural Hazards Partnership (NHP: Hemingway and Gunawan 2018) daily hazard assessment (DHA) wildfire danger rating offering is based on an analysis of output from the Met Office Forecast Severity Index model (MOFSI), crosschecked against EFFIS and reports and information received from members of the England and Wales Wildfire and Scottish Wildfire Forums. The NHP DHA is produced and made available daily to emergency planners, the emergency services and government land management organisations across the UK. The DHA

Table 1. Summary of fire danger thresholds.

MOFSI fire danger		
Very high		FWI > 17.35
Exceptional		FWI > 52.538
Daily hazard assessment amber warning		
November to May		FFMC > 84 and ISI > 5
June to October		FWI > 17.35 and ISI > 10 and either DMC > 49 or DC > 300
Summary of 99th percentiles: 10th to 90th percentile range of values across the UK		
FFMC	May to June	86.6–88.5
FFMC	July to August	86.8–90.4
ISI	May to June	8.2–14.0
ISI	July to August	7.4–13.1
FWI	May to June	15.3–25.4
FWI	July to August	17.0–35.7

The table shows the range in 99th percentile values across the UK for spring and summer. Maps of the distribution of the percentiles in each season are shown in supplementary material.

wildfire Green-Amber-Yellow-Red levels (updated in June 2020) are based on threshold values for the FFMC, ISI, DMC, DC and FWI components (table 1). The thresholds and individual components used vary seasonally (November to May and June to October) due to variations in the amount of combustible material. An ‘amber’ warning level means that severe wildfire conditions are forecast with a likelihood of difficult-to-control wildfires. The ‘red’ level means extreme wildfire conditions are forecast, and occurs when the amber thresholds are passed and other weather conditions are conducive to fire (persistent high temperatures, low humidity and high windspeeds).

de Jong *et al* (2016) argued that the use of the same thresholds everywhere led to an underestimation of wildfire danger in cooler parts of the UK. They concluded that thresholds based on percentiles gave a better indication of fire danger, and recommended using the 99th percentiles of the FWI and FFMC (1% of days have values above the 99th percentile). Maps of seasonal values for the 99th percentiles are presented in supplementary material (available online at stacks.iop.org/ERL/16/044027/mmedia), and table 1 summarises the range in values across the UK. Note that the MOFSI threshold for very high danger is below the 99th percentile FWI across most of the UK—apart from in summer—but the MOFSI exceptional threshold is well above the 99th percentile. The DHA amber thresholds are all below the 99th percentiles for the constituent parts.

This study implemented the Canadian system (using FORTRAN code provided in Van Wagner and Pickett (1985)) as an approximation to MOFSI, with two minor modifications. First, the Canadian day lengths hard-coded into the DMC algorithm were replaced with values more suitable for the UK. Second, the DC algorithm uses a Canadian

approximation to the Thornthwaite formula to estimate potential evaporation: this was replaced by the more physically-realistic Penman–Monteith formula, but the effect is small. The version implemented here uses daily input data, but the Met Office implementation used for fire danger warning uses finer resolution data from the operational weather forecast model.

Six indicators of fire danger are calculated for each cell in the 12 × 12 km observational grid: days with a MOFSI ‘very high or exceptional’, or ‘exceptional’ warning, days with an ‘amber’ level DHA warning, and days greater than the reference period 99th percentile FWI, FFMC and ISI (calculated separately for each grid cell and season). Each indicator is expressed as an average annual value over a 30 year period, and as the likelihood (calculated over 30 years) of having more days than the 1981–2010 mean. This set of indicators was used because, as outlined above, wildfire danger in the UK is not necessarily directly related to the headline FWI indicator.

2.3. Reference climate data

Observed climate data were taken from HadUK-Grid 12 km resolution observational data set (Met Office 2018), supplemented by ERA5 reanalysis (Copernicus Climate Change Service 2017). The HadUK-Grid 12 km data set includes daily minimum and maximum temperature and rainfall up to 2018, but sunshine hours, windspeed and relative humidity are only available as monthly averages. Daily windspeed and relative humidity was therefore estimated from the ERA5 reanalysis, rescaling the ERA5 reanalysis so that the monthly mean equalled the HadUK-Grid monthly mean. Sensitivity studies showed that it was particularly important to use daily windspeed rather than monthly mean windspeed, as this had a big effect on the ISI code and therefore FWI. The time period 1981–2010 is used to represent current climate.

Table 2. Increases in global mean temperature with the RCP2.6 and RCP8.5 projections.

	Increase in temperature above pre-industrial levels (°C)	
	RCP2.6	RCP8.5
2050s	1.6 (1.1–2.2)	2.3 (1.7–2.9)
2100	1.9 (1.3–2.6)	5.1 (4.0–6.5)

The table shows the median estimate of increase in global mean temperature, with the 10th to 90th percentile range in brackets. The average global temperature over the period 1981–2010 was approximately 0.61 °C warmer than pre-industrial levels.

2.4. Climate projections and their application

The UKCP18 land climate projections (Lowe *et al* 2018, Murphy *et al* 2019) consist of four strands: global, regional, local and probabilistic. The global strand consists of an ensemble of 15 climate projections at a spatial resolution of 60 × 60 km made using variants of the HadGEM3 climate model and an ensemble of 12 projections made using CMIP5 climate models. Projections are made for two levels of climate forcing, representing high (RCP8.5) and low (RCP2.6) emissions (table 2). These projections each maintain realistic physical relationships between climate variables and coherent patterns of change across the UK. The regional strand is based on higher-resolution versions of HadGEM3, and in practice gives similar results to the global HadGEM3 projections. The local strand is a smaller number of even higher-resolution projections, and these were not applied. The probabilistic strand consists of 3000 equally-plausible projections at four different levels of forcing, but these do not maintain realistic physical relationships between variables. Most significantly, they can generate unrealistic combinations of vapour pressure and temperature and therefore overestimate the potential range in change in relative humidity—an important driver of fire danger.

The study focused on the global strand climate projections of monthly climate. Figure 2 summarises the regional average summer (June to August) changes in average temperature, rainfall, vapour pressure, relative humidity and windspeed (regions are shown in figure 3), and table 3 shows national average changes in summer temperature and rainfall. The general direction of change is similar across all projections—increase in temperature, reduction in rainfall, reduction in relative humidity, and reduction in windspeed—but there is a clear difference between the two ensembles. The HadGEM3 ensemble tends to generate more extreme changes than the CMIP5 ensemble.

The climate projections to 2100 were applied to the gridded observed 1981–2010 daily time series using the delta method, matching each observed 12 × 12 km grid cell with its corresponding 60 × 60 km climate projection grid cell. Each variable

for a given climate projection was first expressed as an anomaly from its 1981–2010 monthly mean (absolute for temperature and net radiation, relative for the other variables). The resulting time series of monthly anomalies were then smoothed using a 31 year running mean to remove the effect of year to year variability and extract the climate change signal. In order to calculate anomalies for the last 15 years of the projections the anomaly time series were extrapolated using linear regression. There can be large differences in anomaly from one month to the next—which introduces unrealistic steps at month boundaries—so the monthly anomalies were interpolated to the daily scale before being applied to the observed daily data. Time series from 2011 to 2100 were constructed by first repeating the 1981–2010 time series three times to create a long unperturbed record and then applying the annual time series of anomalies (see supplementary material).

The delta method assumes no change in relative climate variability from year to year, and that the proportional change in a variable does not vary with the magnitude of that variable. It is possible that climate change would generate increased variability in summer temperatures from year to year, and this would increase the number of high temperature extremes. A reduction in the number of wet days would increase the chance of prolonged dry spells.

The study uses the delta method to apply changes to observed daily time series rather than apply bias adjustment to the monthly UKCP18 projections for three reasons. First, observed data is used to characterise the current climate because this observed experience is familiar to stakeholders. Second, different bias adjustment approaches exist correcting for different aspects of bias, and all assume that the adjustments continue into the future. Third, it would have been impractical to test and apply bias adjustment methods which preserved relationships between variables for all projections and locations.

2.5. Regional averages and example sites

The fire danger model is implemented at a spatial resolution of 12 × 12 km across the UK. Regional averages are also calculated for UK regions (9 in England, 3 in Scotland, and Wales and Northern Ireland: figure 3), weighting the grid cell values by the area of land under improved grassland, heathland, bog, marsh and grassland taken from the 2015 UK Land Cover Map (CEH 2017). These are the land covers with the greatest number of wildfires (de Jong *et al* 2016).

Six example locations (table 4 and figure 3) are used to illustrate year to year variability in fire danger and to explore the relative importance of the different drivers of fire danger. These locations are representative of different meteorological conditions across the UK, and all are exposed to fire danger.

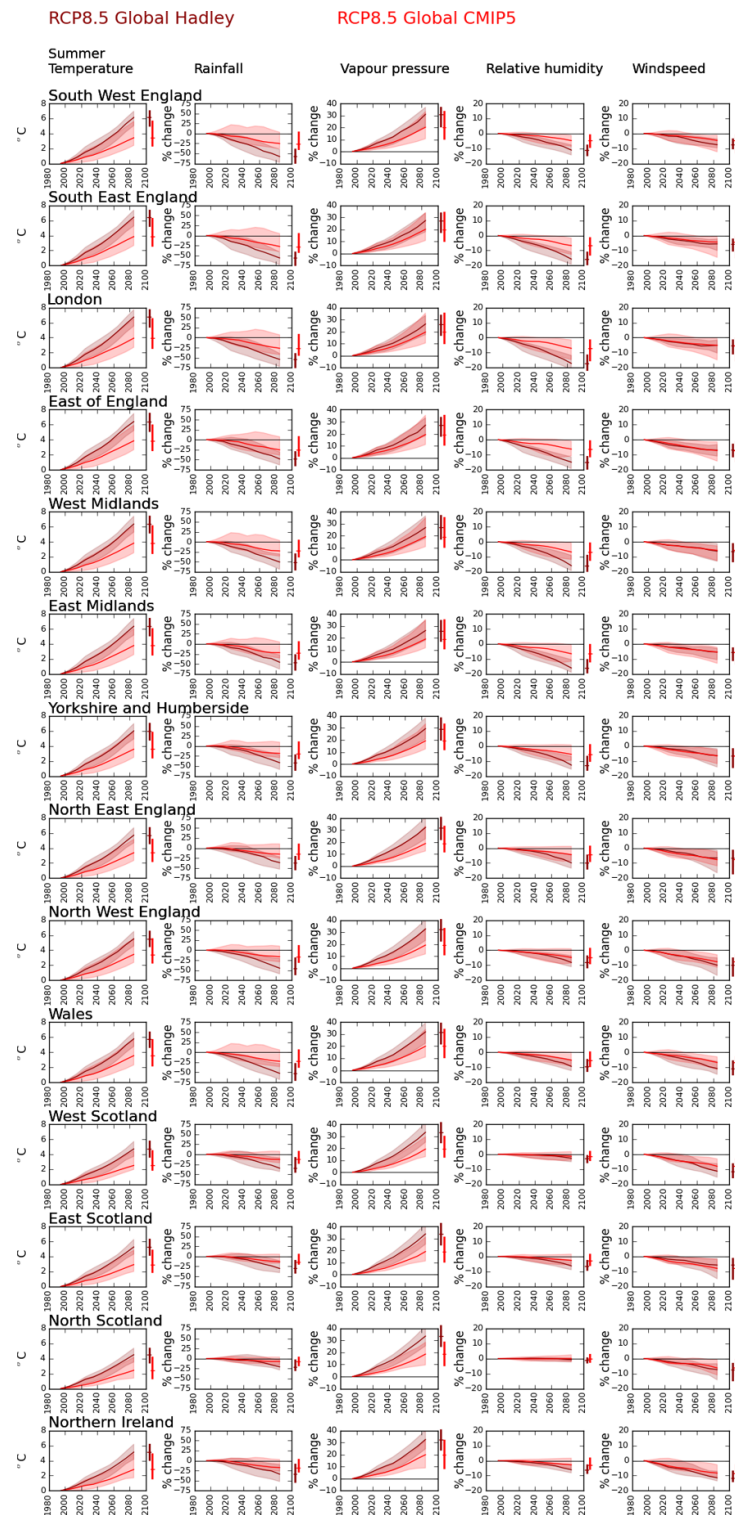


Figure 2. Regional average changes in 30 year mean summer climate, with the RCP8.5 emissions projection. The plots show the median and range across the HadGEM3 and CMIP5 ensembles separately, and values are plotted at the centre point of the 30 year period.

3. Current exposure to wildfire danger

Figure 4 shows the 1981–2010 mean number of days per year for the DHA and MOFSI warning indicators (note that, by definition, there will be on average 3.65 d yr^{-1} greater than the 99th percentile threshold indicators). The greatest average number of

days above the DHA threshold is in south and east England, reaching a maximum of up to 20. There are far fewer days above the MOFSI exceptional warning threshold, with maxima across the East Midlands.

Time series from 1981 to 2018 are shown in figure 5 for the six sample sites: at none of these sites were there days with MOFSI exceptional warnings



between 1981 and 2018. The large year to year variability in number of days with fire danger is evident, and—over this short period—there is little evidence of trend.

Figures 4 and 5 show fire danger over space and time, but do not directly map onto observed fire experience between 1981 and 2018 and figure 4 is not to be interpreted as a map of fire risk. This is because fires need a source of ignition. These are rare, so actual fires at a place are rare, and fires do not necessarily occur in places or years with the highest

fire danger. This makes it difficult to validate both the time series and spatial pattern of fire danger. In practice, most fires in the UK are located close to the rural–urban fringe (Gazzard *et al* 2016) and the distribution of actual fires across space is therefore much more closely related to land cover than the climatic factors affecting danger. The distribution across the UK of the largest fires—shown using EFFIS data aggregated to the 12×12 km resolution in figure 4—is also closely linked to land cover and land management practices.

Table 3. National average change in summer (June–August) temperature and rainfall across the UK, relative to the 1981–2010 average.

	2050s						2080s					
	HadGEM			CMIP5			HadGEM			CMIP5		
	Median	Low	High	Median	Low	High	Median	Low	High	Median	Low	High
Summer temperature (°C)												
England	3.47	2.90	3.89	2.10	1.36	2.87	6.01	5.25	6.77	3.65	2.81	5.45
Wales	3.11	2.67	3.70	1.99	1.35	2.69	5.61	4.84	6.43	3.55	2.57	5.32
Scotland	2.70	2.15	3.18	1.51	0.98	2.56	4.77	4.02	5.59	2.59	2.16	4.33
Northern	2.76	2.38	3.40	1.49	1.18	2.59	5.04	4.23	5.93	2.82	2.00	4.38
Summer rainfall (% change)												
England	−29	−39	−16	−14	−22	10	−54	−63	−34	−21	−29	−2
Wales	−30	−41	−16	−14	−26	8	−57	−66	−38	−23	−36	1
Scotland	−13	−18	−2	−7	−11	2	−29	−33	−13	−10	−16	1
Northern	−19	−23	−8	−12	−17	4	−36	−48	−21	−17	−26	2

The table shows the median across the ensemble members, together with the lowest and highest values.

Table 4. Locations of the six sample sites.

Location	Region	Longitude and latitude	Landscape type
Swinley	South East England	0.75° W, 51.4° N	Lowland heath and forest on rural-urban fringe
Dorset Heaths	South West England	2° W, 50.7° N	Lowland heath
Saddleshworth	North West England	2° W, 53.5° N	Upland moorland
Knockando	North East Scotland	3.4° W, 57.5° N	Upland heathland
Phenzhopehaugh	Southern Scotland	3° W, 55.4° N	Upland heathland
Tonypandy	South Wales	3.6° W, 51.6° N	Upland moorland

4. Change in fire danger across the UK

4.1. Change in danger with high emissions

Figures 6 and 7 show regional average values of the six indicators with the RCP8.5 high emissions projections, expressing each indicator in terms of the average annual number of days and likelihood of experiencing more than the mean number of days (results are provided in supplementary material). The HadGEM3 and CMIP5 ensembles are plotted separately, and each are represented by the median plus the range between the second lowest and second highest. Figure 4 shows the average spatial pattern of change in the six indicators in the 2050s and 2080s, calculated as the median across the 15 HadGEM3 projections. Table 5 summarises results by region for the 2050s and 2080s with the HadGEM3 projections (results for the CMIP5 ensemble, and for other years, are given in supplementary material). In the reference period the likelihood of experiencing more than the mean number of days is well below 50% for each indicator (and especially the MOFSI exceptional danger indicator) because the annual distribution of danger days is highly skewed. Several conclusions can be drawn from the plots.

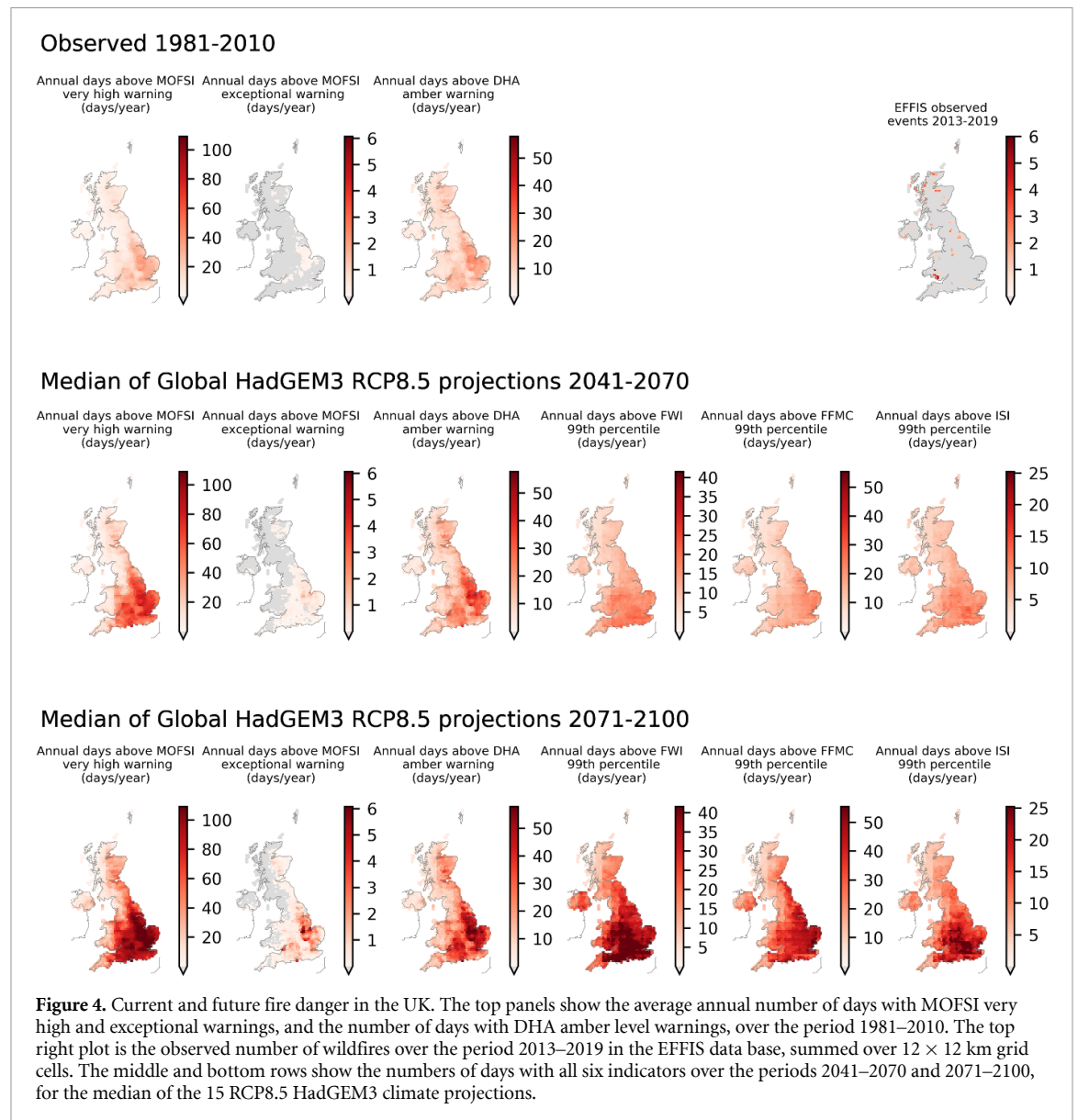
There is a large difference between the two ensembles of projected change in climate. The increase in fire danger is much greater with the HadGEM3 ensemble than the CMIP5 ensemble, consistent with the differences in change in climate between the two ensembles (figure 2). The HadGEM3 ensemble members produce greater temperature

increases, greater reductions in relative humidity, and larger reductions in spring and summer rainfall than the CMIP5 ensemble, so produce larger changes in fire danger. There is also a large uncertainty range across each ensemble (section 5.1).

The projections imply a very large increase in fire danger across the UK, particularly in the south, east and central regions of England, but the changes depend on the indicator due to their different sensitivity to change in climate (see section 5.1) and the differences between the absolute and variable (percentile) indicators.

The number of days greater than the MOFSI ‘very high’ threshold is greatest in southern and eastern England (increasing from an average of 20 d yr^{−1} in southern England over the period 1981–2010 to up to 70 by the 2050s and up to 111 by the 2080s). The likelihood of experiencing more than the current mean number of days rises from 37% to up to 98% by the 2050s and 100% by the 2080s. Further north in England the numbers of days only begin to increase substantially by the 2040s, and in west and north Scotland and Northern Ireland change little through the century in absolute terms, but the likelihood of experiencing more than the current mean increases (from 27% now to up to 64% by the 2080s in northern Scotland).

The number of days above the MOFSI ‘exceptional’ threshold remains low through the 21st century in northern England, Wales, Scotland and Northern Ireland, but increases substantially after the 2050s in southern and, particularly, eastern England



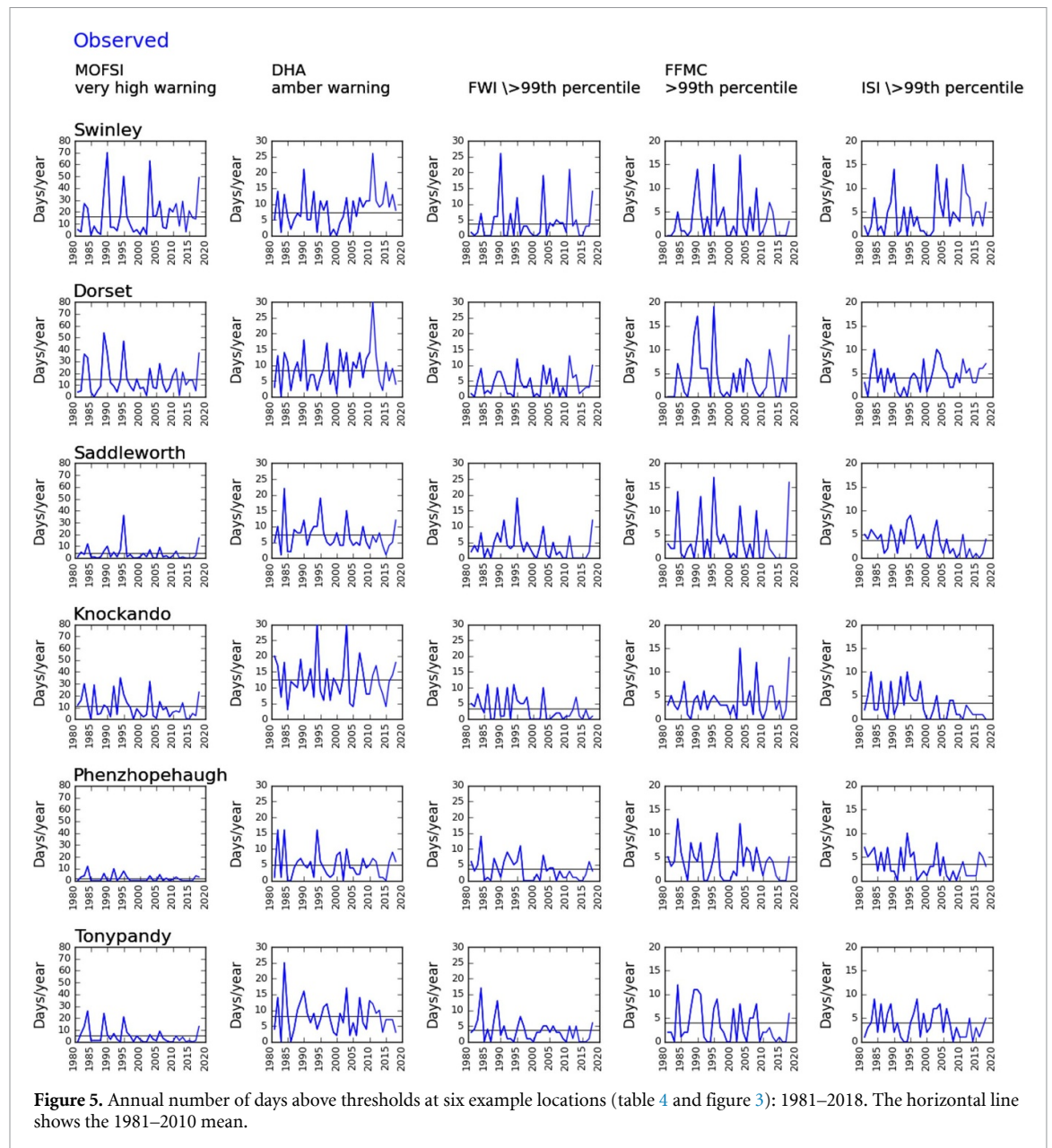
(up to an average of 3 d per year by the 2080s and a likelihood of 68%).

The number of days above the DHA amber warning threshold increases across more of the UK, but increases are considerably higher in the south and east. By the 2050s there could be more than 30 d per year with DHA amber warnings across south central England, compared with around 10–15 now, and at least an 80% chance of experiencing more than the current mean. In eastern Scotland the mean increases from 9 to up to 16 d per year by the 2050s, and the chance of having more than the mean increases to between 66% and 76%.

Although the reference period danger is (by definition) the same for the three percentile indicators and constant across the UK, figures 6 and 7 show differences in changes between the indicators and across space. The spatial variability reflects variability in the change in climate across the UK—bigger increases in temperature, bigger reductions

in relative humidity and bigger reductions in rainfall in the south—and the differences between the FWI, FFMCI and ISI indicators reflects their different sensitivity to different aspects of climate change (see section 5.1). FFMCI is relatively insensitive to changes in rainfall but reductions in rainfall lead to increases in FWI. The greatest change in the FWI indicator is therefore in areas with the greatest reductions in rainfall, so the largest increase is therefore in the south rather than the east of England. ISI is more directly affected by changes in windspeed than the other indicators, and the projected reductions in windspeed explain why this indicator increases less rapidly than the others.

The MOFSI and DHA indicators are based on absolute, rather than spatially-variable percentile thresholds. The spatial variability in these indicators is therefore more determined by variability in the reference conditions—specifically how often the



thresholds are currently exceeded—than by variability in climate change.

4.2. Seasonal distribution of change in fire danger

More fires currently occur in spring than summer, and few fires occur during winter and autumn (de Jong *et al* 2016, Glaves *et al* 2020). Figure 8 shows change in seasonal fire danger across the UK for the six indicators (note that the axis limits vary between regions because the purpose of the plot is to show differences between seasons, not regions). MOFSI very high and exceptional warnings currently occur almost exclusively in summer and summer sees the greatest increase, but in the east of England fire danger increases too in autumn. There is little apparent increase in spring fire danger with the MOFSI exceptional warning indicator. In contrast, the DHA amber warning indicator has the greatest number of danger days in spring: this is primarily because the criteria

used to define an amber warning change between May and June. The number of danger days in summer, however, increases more rapidly than the number of danger days in spring. The three percentile indicators show the greatest increase in fire danger in summer and autumn, with much smaller increases in spring. This suggests that the increase in fire risk (a product of chance of ignition and fire danger) may be smaller than the increase in annual fire danger.

4.3. Effect of reducing emissions on fire danger

Figure 9 compares the annual fire danger indicators under high (RCP8.5) and low (RCP2.6) emissions. It demonstrates that reducing emissions to a level consistent with an increase in global mean temperature of around 2 °C reduces very substantially the increase in fire danger, but does not eliminate it. With this low emissions scenario danger increases most in south

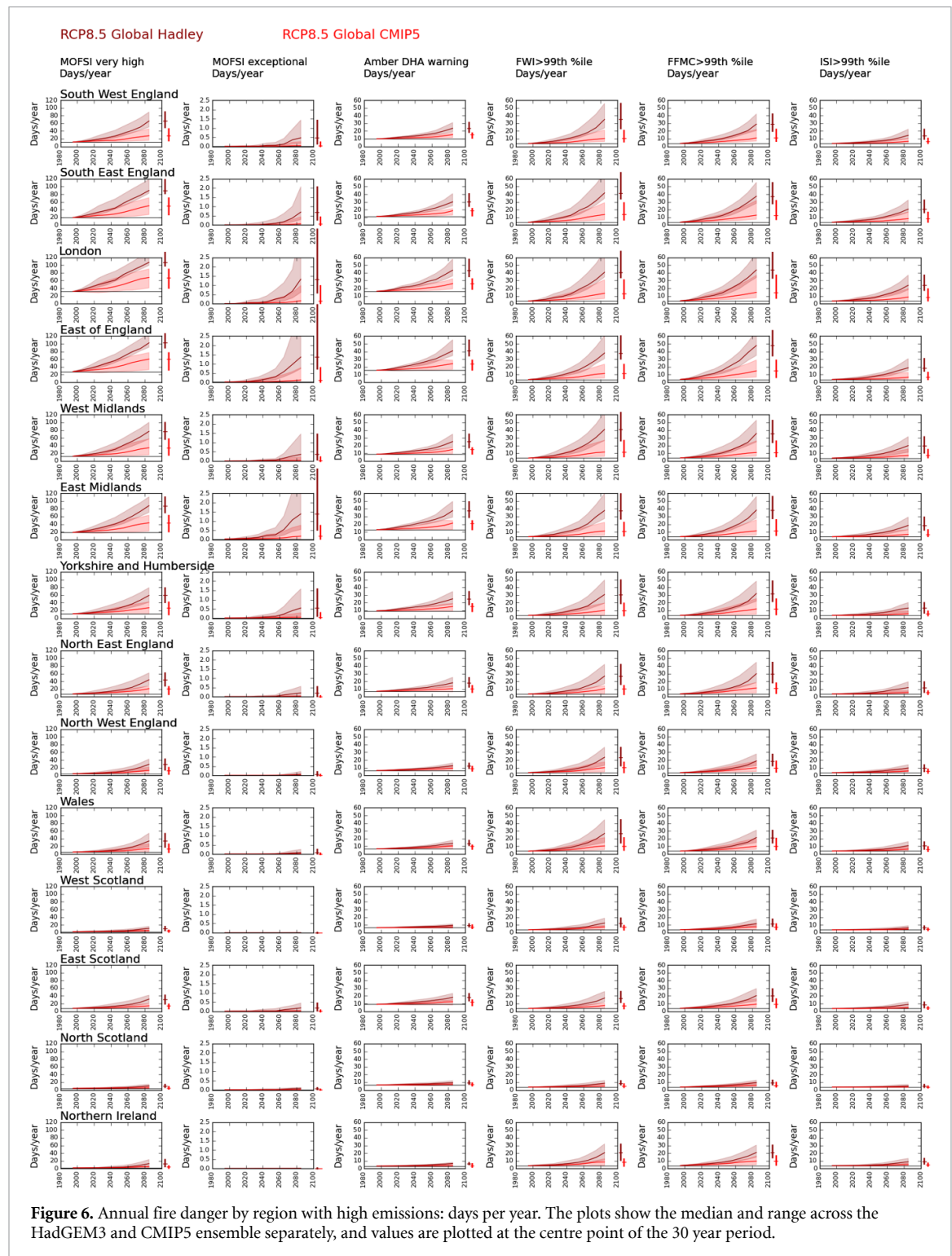


Figure 6. Annual fire danger by region with high emissions: days per year. The plots show the median and range across the HadGEM3 and CMIP5 ensemble separately, and values are plotted at the centre point of the 30 year period.

east and eastern England, but is little changed across northern England, Wales, Scotland and Northern Ireland.

5. Drivers of change in fire danger

5.1. Relative importance of different dimensions of climate change and sources of uncertainty

The changes in fire danger discussed in the previous section are driven by changes in temperature, relative

humidity, rainfall, windspeed and—to a much lesser extent via potential evaporation—net radiation. Figure 10 shows the relative importance of these different drivers for each indicator by region over the period 2071–2100, with the HadGEM3 ensemble. Each panel shows the effect of (a) just change in temperature (holding rainfall, relative humidity, windspeed and net radiation constant at 1981–2010 levels), (b) change in temperature and rainfall, (c) change in temperature and

Table 5. Regional average values of the six fire danger indicators over 1981–2010, 2041–2070 and 2071–2080, with the HadGEM3 RCP8.5 ensemble. The annual % chance shows the likelihood of experiencing more days than the reference period mean.

	D yr ⁻¹						Annual % chance					
	2050s			2080s			2050s			2080s		
	1981–2010	Mid	Low	High	Mid	Low	High	1981–2010	Mid	Low	High	1981–2010
MOFSI very high danger												
South West England	12	33	29	45	62	53	83	29	77	74	89	93
South East England	20	55	51	70	89	84	111	35	92	91	99	100
London	32	74	70	90	106	102	124	41	92	90	100	100
East of England	28	66	61	82	102	92	121	36	94	93	99	100
West Midlands	13	37	32	58	75	58	96	30	86	79	99	100
East Midlands	19	47	44	67	86	71	106	34	92	88	100	100
Yorkshire and Humberside	11	28	23	44	58	42	78	34	75	68	94	100
North East England	8	19	16	33	40	29	61	33	70	62	89	100
North West England	5	10	9	18	28	15	41	27	56	49	73	91
Wales	5	12	11	22	32	21	53	25	54	49	82	98
East Scotland	8	15	13	23	29	21	38	32	62	53	80	93
West Scotland	3	5	4	8	10	7	16	25	39	35	57	76
North Scotland	4	6	5	9	10	8	13	27	41	35	52	64
Northern Ireland	2	4	3	7	11	8	20	19	41	36	61	90
MOFSI exceptional danger												
South West England	0	0.06	0.04	0.14	0.41	0.2	1.25	0.1	3	2	6	9
South East England	0.01	0.09	0.06	0.25	0.57	0.35	1.88	0.0	6	4	14	18
London	0.01	0.24	0.14	0.68	1.23	0.76	3.81	0.0	14	8	30	34
East of England	0.02	0.29	0.16	0.69	1.36	0.86	3.55	0.0	13	8	26	30
West Midlands	0	0.04	0.03	0.13	0.34	0.13	1.25	0.1	3	2	7	8
East Midlands	0.03	0.26	0.16	0.64	1.34	0.86	3.23	0.3	10	6	21	25
Yorkshire and Humberside	0.01	0.09	0.04	0.25	0.53	0.19	1.55	0.0	5	2	12	9
North East England	0	0.03	0.01	0.1	0.19	0.05	0.54	0.0	2	0.4	7	4
North West England	0	0	0	0.02	0.05	0.01	0.21	0.0	0.4	0.2	1	1
Wales	0	0.01	0.01	0.03	0.06	0.03	0.24	0.1	0.4	0.3	2	2
East Scotland	0	0.02	0.01	0.07	0.16	0.05	0.36	0.1	1.1	0.6	4	3
West Scotland	0	0	0	0	0	0	0.01	0.0	0.0	0.0	0	0
North Scotland	0.01	0.02	0.01	0.04	0.06	0.03	0.12	0.1	0.6	0.4	2	1
Northern Ireland	0	0	0	0.01	0.01	0.01	0.03	0.0	0.4	0.3	1	1

Table 5. (Continued.)

	D yr ⁻¹						Annual % chance							
	2050s			2080s			1981–2010	2050s			2080s			
	1981–2010	Mid	Low	High	Mid	Low		High	Mid	Low	High	Mid	Low	High
DHA amber danger	10	16	14	18	23	19	30	40	82	80	91	97	95	100
	11	19	17	22	30	26	41	37	91	89	98	100	100	100
	16	28	25	33	43	38	57	36	89	86	100	100	100	100
	16	27	24	31	41	36	54	37	92	89	98	100	100	100
	9	15	14	18	24	20	32	39	86	80	98	100	98	100
	13	23	21	27	36	32	47	39	89	84	99	100	100	100
	10	16	15	20	25	22	33	39	79	70	94	99	93	100
	7	12	11	14	18	16	24	37	75	67	89	97	88	100
	6	9	9	11	12	11	16	38	68	59	81	91	83	96
	7	10	9	11	14	14	18	38	70	61	84	94	87	98
	9	13	12	16	18	18	22	41	71	63	83	89	82	96
	6	8	7	9	9	9	11	40	58	54	70	77	69	86
FWI 99th percentile	6	8	7	9	9	9	12	41	52	46	63	66	58	76
	3	5	4	5	6	6	8	39	63	56	76	86	79	94
	4	14	12	21	32	26	52	28	79	73	91	97	94	100
	4	18	14	26	39	35	64	26	82	79	94	99	98	100
	4	18	15	27	40	36	63	28	82	74	93	97	96	100
	4	16	13	24	35	31	60	29	82	75	92	99	97	100
	4	15	13	25	38	27	62	26	87	79	97	100	98	100
	4	14	12	24	37	27	59	26	84	79	96	99	98	100
	4	11	9	20	31	20	48	28	78	72	91	96	92	100
	4	10	8	19	23	18	41	27	79	69	89	94	89	99
	4	9	8	14	22	14	34	32	70	63	83	91	85	96
	4	10	9	17	26	19	44	30	70	67	89	95	91	98
East of England	4	8	7	13	16	12	24	32	66	58	80	87	80	95
	4	7	6	10	12	10	17	34	54	51	70	76	69	87
	4	5	5	7	8	7	11	32	47	44	56	64	57	75
	4	8	7	13	17	13	30	32	58	52	78	90	77	96

(Continued)

Table 5. (Continued.)

	D yr ⁻¹										Annual % chance				
	2050s					2080s					2050s				
	1981–2010	2050s			High	Mid	2080s			High	1981–2010	Mid	2050s		
		Mid	Low	High			Mid	Low	High				Low	Mid	High
FFMC 99th percentile															
South West England	4	13	12	19	39	27	22	22	39	30	30	79	76	96	98
South East England	4	17	14	23	54	36	30	30	54	30	30	84	79	99	100
London	4	19	16	27	63	42	37	37	63	28	28	85	80	100	100
East of England	4	21	17	28	67	45	37	37	67	30	30	91	85	100	100
West Midlands	4	14	12	21	51	32	26	26	51	29	29	83	75	100	100
East Midlands	4	15	13	24	55	37	30	30	55	29	29	84	78	100	100
Yorkshire and Humberside	4	13	11	21	44	27	24	24	44	32	32	80	73	98	100
North East England	4	12	10	18	39	23	19	19	39	32	32	75	70	94	100
North West England	4	9	8	13	24	16	14	14	24	29	29	67	58	90	96
Wales	4	10	9	14	28	19	15	15	28	34	34	73	67	94	96
East Scotland	4	10	8	14	26	18	14	14	26	31	31	74	66	93	98
West Scotland	4	7	6	9	16	11	9	9	16	31	31	59	52	80	88
North Scotland	4	6	6	7	12	9	8	8	12	28	28	53	48	71	78
Northern Ireland	4	10	8	15	28	19	15	15	28	32	32	73	64	94	98
ISI 99th percentile															
South West England	4	7	6	10	20	14	11	11	20	33	33	64	56	85	92
South East England	4	10	8	14	31	20	18	18	31	32	32	73	67	92	97
London	4	11	9	16	37	23	20	20	37	33	33	74	67	94	99
East of England	4	9	8	13	29	18	16	16	29	33	33	79	69	96	100
West Midlands	4	9	7	12	30	19	13	13	30	31	31	77	62	96	99
East Midlands	4	8	7	12	26	17	14	14	26	29	29	77	66	98	100
Yorkshire and Humberside	4	7	5	8	18	12	9	9	18	31	31	67	49	90	97
North East England	4	6	5	9	17	12	7	7	17	32	32	57	46	88	97
North West England	4	6	5	7	13	9	6	6	13	33	33	56	45	77	87
Wales	4	6	5	8	15	10	7	7	15	32	32	55	47	80	88
East Scotland	4	5	4	7	12	9	6	6	12	32	32	51	39	77	86
West Scotland	4	4	4	6	8	6	5	5	8	32	32	40	35	58	69
North Scotland	4	4	4	5	6	5	5	5	6	32	32	39	32	47	59
Northern Ireland	4	5	5	7	11	8	7	7	11	29	29	43	39	69	85

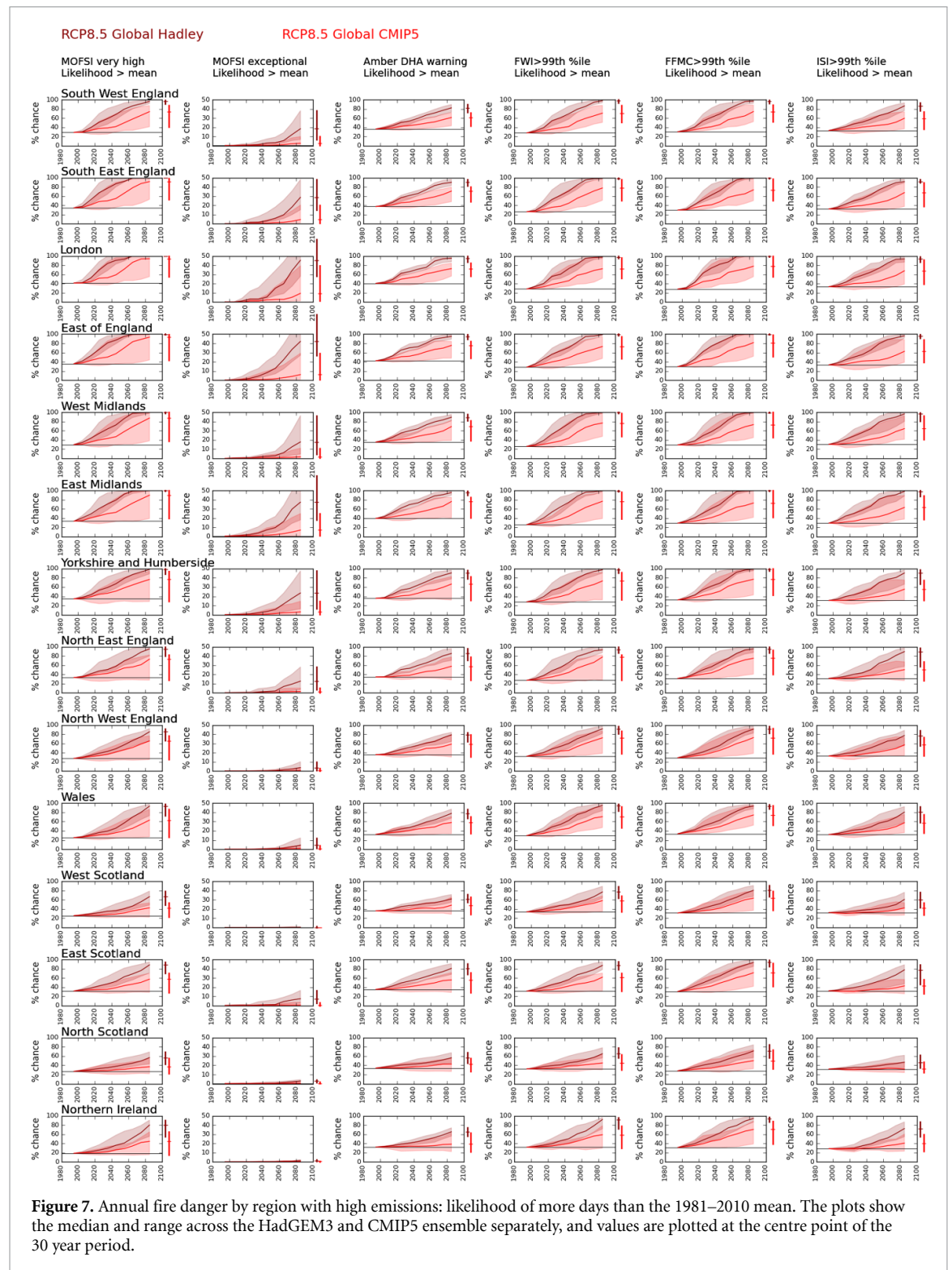


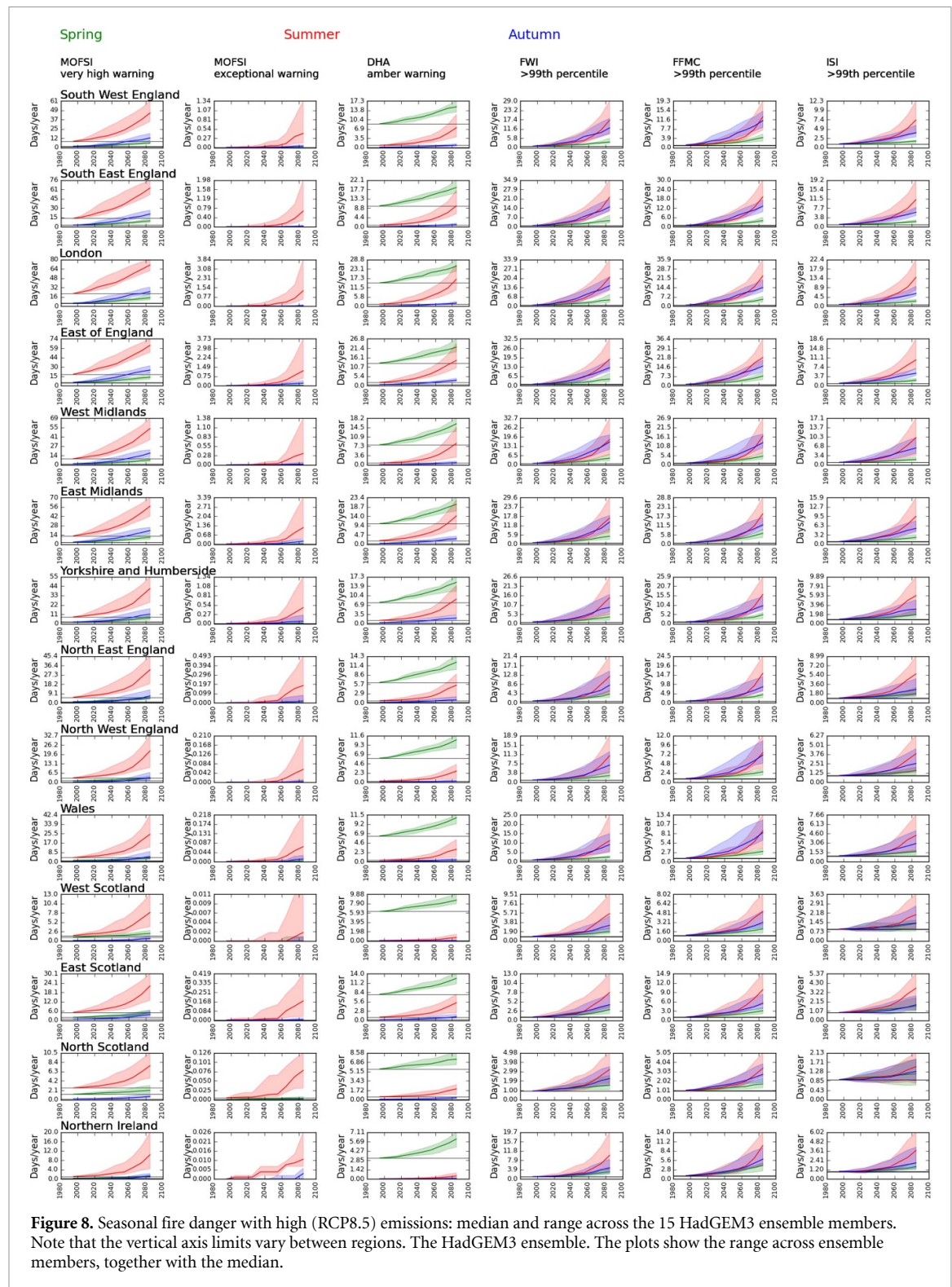
Figure 7. Annual fire danger by region with high emissions: likelihood of more days than the 1981–2010 mean. The plots show the median and range across the HadGEM3 and CMIP5 ensemble separately, and values are plotted at the centre point of the 30 year period.

vapour pressure, (d) change in temperature, vapour pressure and rainfall, and finally (e) change in all variables.

Just under half of the change in the FFMCI percentile indicator is due to the increase in temperature, and just over half is due to the reduction in relative humidity (with the importance of relative humidity less further north where the changes are smaller). Change in rainfall has very little effect on change in

FFMCI. Most of the uncertainty in change in FFMCI is due to uncertainty in change in relative humidity.

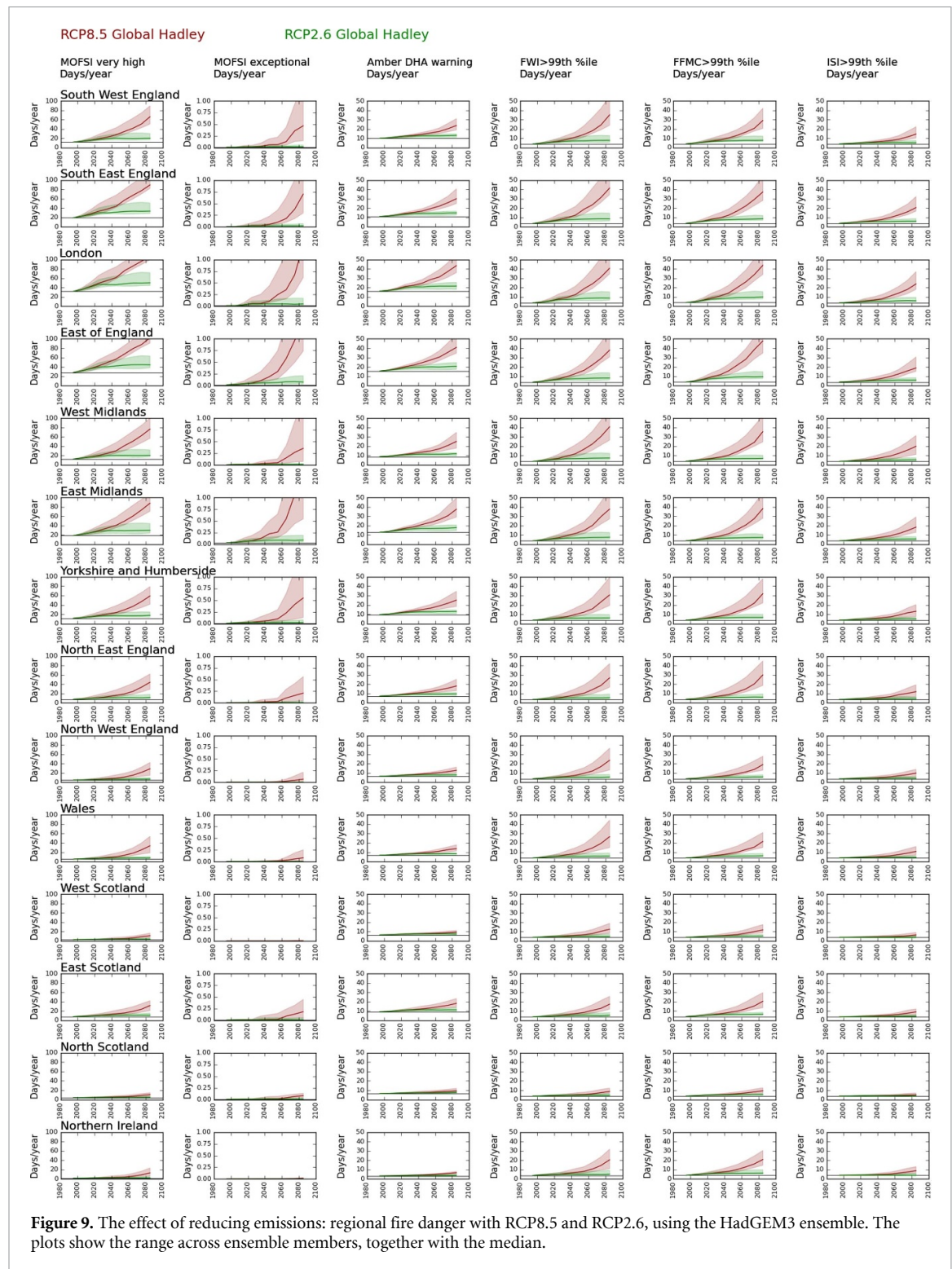
Change in rainfall has a greater effect on the ISI percentile indicator, although is still small compared to the effects of change in relative humidity. ISI includes windspeed, so the general reduction in windspeed offsets the effect of changes in the other drivers. The uncertainty range is again largely driven by uncertainty in the change in relative humidity.



FWI combines FFMCI and ISI, with other components (figure 1). The sensitivity to windspeed is less than for ISI, and the effect of changes in rainfall are increased compared to FFMCI. The rainfall and relative humidity changes have a similar magnitude effect—except in the north where relative humidity changes are small—and the uncertainty in changes in FWI are therefore driven by both uncertainty

in rainfall and relative humidity. The two are not well correlated across the ensemble members, so the uncertainty range in FWI is large.

The effects of changes in different climate variables on the MOFSI indicator is similarly to effects on FFMCI, but numerically different because MOFSI is based on absolute thresholds which are correspond to different percentiles in different parts of the UK.



Uncertainty in the MOFSI exceptional danger level is particularly wide, and this is due to the combined effect of uncertainty in humidity and rainfall changes.

The amber DHA warning is based on combinations of the component codes, but sensitivity to change and the major sources of uncertainty are similar to those for the FFMCI indicator.

Changes in fire danger are therefore a function of increases in temperature, reductions in relative

humidity and, to a lesser extent, reductions in rainfall. For a given ensemble, uncertainty in relative humidity and rainfall changes have the largest effect on uncertainty in change in fire danger. The difference between the HadGEM3 and CMIP5 ensembles is partly due to the greater increase in temperature with the HadGEM3 ensembles, but partly due also to the greater reduction in relative humidity and, at least in some seasons, rainfall.



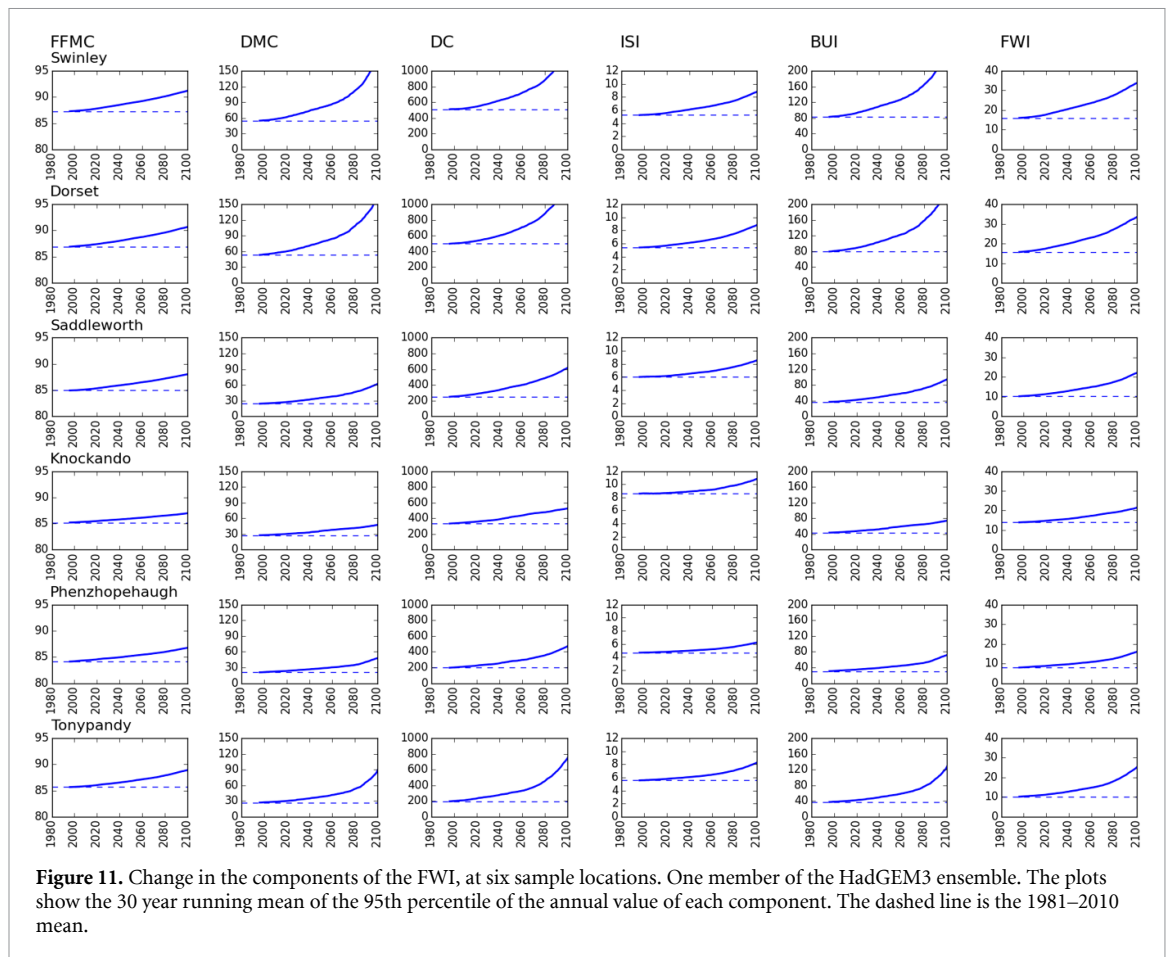
Figure 10. The relative importance of changes in temperature, rainfall, relative humidity and windspeed on indicators of fire danger, by region over the period 2071–2100 with the HadGEM3 RCP8.5 ensemble. The plots show the median and range: the grey dashed lines show the 1981–2010 mean. Note that the axis limits vary between regions.

5.2. Changes in different components of fire danger

The FWI indicator is built up from five components (figure 1). Figure 11 shows the percentage change in these components, plus the FWI, for the six sample locations, for one member of the HadGEM3 ensemble. Each plot shows the annual 95th percentile of each component, calculated as a 30 year running mean. Note that the final FWI indicator is not a

simple average of the five components, as each is given a different weight.

All the components of the FWI increase through time, and therefore all contribute to the increase in FWI. The proportional change in 95th percentile value of FFMCI is small, but this component varies relatively little from year to year and small changes correspond to large changes in the



frequency with which a given absolute value is passed.

6. Conclusions and implications

This paper has presented an evaluation of change in fire danger across the UK due to climate change. Fire risk is a function of danger and the chance of ignition, and given that the chance of ignition varies differently over space and through the year to fire danger, changes in risk will not necessarily follow exactly changes in danger. However, the results give an indication of how fire hazard in the UK will change over the 21st century.

The results show that fire danger increases across the whole of the UK, but the extent and geographical variability in change depends on the specific indicator of fire danger. The absolute danger now and into the future is greatest in the south and east of England, but danger increases too further north. The increase in fire danger is greatest in summer but this is not necessarily when the risk is highest due to the timing of sources of ignition. Reducing emissions to levels consistent with achieving international climate policy targets reduces significantly the increase in fire danger, but does not eliminate it.

There is a wide uncertainty range in how fire danger increases through the 21st century due to

variability across plausible climate scenarios. Approximately half of the increase is due to increases in temperature and most of the rest is due to reductions in relative humidity. There is a large difference between the projections of change in danger between the two ensembles of climate projections, and the HadGEM3 ensemble generates larger increases in danger than the CMIP5 ensemble. This is because the HadGEM3 ensemble not only produces larger increases in temperature, it also produces greater reductions in relative humidity. The uncertainty in projected changes in fire danger is therefore determined by, in approximately equal proportions, by the combined effects of uncertainty in change in temperature, relative humidity and rainfall (uncertainty in rainfall has very little effect on the FFM indicator).

There are several implications of these results. First, the projected large increase in wildfire danger across the UK implies that the risk should be given increased attention not only in emergency planning (such as in the National Security Risk Assessment and National Risk Register, Local Resilience Forum's Community Risk Registers and Fire and Rescue Service's Integrated Risk Management Planning) but also in land use spatial planning and management (infrastructure, development, agriculture, conservation, forestry etc). Second, greater caution and more stringent guidelines are needed for planning

activities which can trigger wildfires. Controlled and prescribed burning is an important component of lowland and upland land management, for example, and changes in the risk of a fire spreading could influence practices. Third, the different indicators of danger give subtly different indications of where and how danger will change, and this implies a need for the further development of fire danger indices and indicators more finely tuned to UK fuel sources and meteorological conditions. Fourth, the results show change in fire danger: change in wildfire risk (in terms of area burnt, for example), will be strongly influenced by where fires are started and how the sources and locations of ignition might change. An improved understanding of current and future fire risk therefore needs estimates of change in danger to be combined with models characterising the likelihood of ignition.

Data availability statement

All data that support the findings of this study are included within the article (and any supplementary files).

Acknowledgments

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UK and Global Food Security in the Era Of ‘Permacrises’

The UK's food security is currently dependent on complex international supply chains that are under threat from climate change, conflicts and geopolitical stressors; at the same time, climate change and the repercussions of Brexit challenge the UK's ability to increase self-reliance and internal resilience. Jennifer Cole and Ivica Petrikova examine the key drivers and pressures on global food security and how this is likely to be impacted in the near- and mid-term future, including by shifting geopolitical alliances.

Food security is a cornerstone of UK national resilience¹ but it is vulnerable to external supply shocks,² and is likely to come under increasing threat from climate change in the mid- to long-term future.³ UK Government documents such as *Global Britain in a Competitive Age: The Integrated Review of Security, Defence, Development and Foreign Policy*,⁴ the Development, Concepts and Doctrine Centre's 'Global Strategic Trends',⁵ and the Environment, Food and Rural Affairs Committee's recent report 'Food Security'⁶ articulate the complex links between not only UK national security and food

security, but also global food security and its influence on UK food availability, export opportunities and associated global trade. The precarity of UK food supply chains, highlighted by RUSI⁷ nearly a decade ago, received renewed attention during the Covid-19 pandemic, as supermarket shelves emptied and increasing numbers of the UK public were driven to rely on food banks when jobs were furloughed on only 80% of normal pay, fuel and energy prices increased, and the precarity of many people's day-to-day finances was exposed. UK food supplies have since become further challenged by the war in

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The global food system is under strain.
Courtesy of zhang yongxin / Adobe Stock

Ukraine and may be impacted by the ongoing Middle Eastern conflicts, as the Red Sea unrest disrupts global food trade.⁸

The UK is currently a net-importer of food – in 2020, it imported 46% of the food it consumed⁹ – and so is reliant on overseas production to feed its population. This reliance is likely to be tested further over the coming decades. The world has now arguably entered an era of ‘permacrisis’,¹⁰ where as soon as we emerge from one crisis, another follows. The climate crisis in particular is expected to have a significant disruptive effect on global food security.¹¹ Combined with the increasing number and intensity of national and international conflicts and natural hazards that disrupt supply chains, it is time to reappraise the challenges and opportunities offered

by the UK’s approach to internal and global food security.

This article analyses the UK’s national and global food security in the ‘permacrisis’ era from the perspective of global geopolitics. In discussing food security, it concentrates predominantly on its macro-level pillars – food availability and food stability.¹² A stable and sufficient supply of food at the country or regional level is not in itself a guarantee for satisfactory food security at the household and individual level: the explosion in the number of food banks in the UK since 2008, as government social welfare declined, illustrates well this point.¹³ Nevertheless, satisfying the micro-level aspects of food security, that is food access and utilisation, is not possible without the macro-level ones on

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UK and Global Food Security in the Era Of 'Permacrisis'

which this article focuses. An understanding of the historical situation, which explains the UK's position as a net food importer, sheds light on global food trade patterns and their potential disruptions. This in turn provokes discussion of potential responses to the disruptions and will help to identify appropriate policy recommendations.

UK Food Production

Britain last produced enough food for its population in 1846, when the Corn Laws, which had protected British farmers from cheaper European grains but led to higher food prices, were repealed. Thereafter, British domestic agricultural production experienced a gradual decline. At the onset of the First World War, the UK was importing 60% of the food it consumed.¹⁴ Blockades and submarine warfare during both world wars prompted the UK to convert large swathes of pastures and parkland into arable land and rapidly expand domestic food production.¹⁵ Nevertheless, during neither conflict did the UK attain food self-sufficiency; at the peak of food production during the Second World War, the UK rationed key food items and still imported 56% of the calories consumed.¹⁶ Today, the UK imports about half the food that it consumes, and this proportion may increase in the coming years.

In a 2010 food security assessment,¹⁷ the Department of Environment, Food and Rural Affairs (DEFRA) tested a scenario in which the UK was cut off from all its trade partners, to assess if the country would be able to feed its population on its own. DEFRA concluded that it would be possible, but only if all arable land were used in crop production

for human rather than livestock consumption: the production of 100 g of protein from grains, pulses and peas requires only 2–5% of the land required to produce the same 100 g from lamb or beef.¹⁸ DEFRA's assumption in the exercise that cereal yield productivity will continue to rise in the future might not hold true, however, as both soil degradation – only 10% of soils in South West England are classified as not degraded¹⁹ – and increased incidence of adverse weather events threaten to reduce future crop yields.

Britain last produced enough food for its population in 1846

At the same time, the UK's post-Brexit agricultural scheme, in combination with new free-trade agreements, might lead to further reduction in domestic UK agricultural production. In 2022, the UK rolled out the Environmental Land Management scheme,²⁰ a replacement of the EU's Common Agricultural Policy, which is gradually substituting direct payments to farmers with payments for environmental protection and restoration of ecosystem services.

The same year, the UK signed a free-trade agreement with Australia, a large agri-food exporter, without any safeguards on environmental or animal welfare standards for food imports, on which tariffs will gradually be removed,²¹ potentially making it cheaper to import food from overseas than to produce it at home. In combination, and more so if similar free-trade agreements are signed in the future, these developments might lead to a situation where UK farmers focus more on environmental

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 21. Compassion in World Farming, 'Safeguarding the UK's Food and Farming Standards in Trade: Lessons from the Australia-UK Free Trade Agreement', 30 March 2022, [REDACTED] accessed 1 February 2024.

protection while reducing domestic agricultural and livestock production.

Globalised Food Systems

The UK's substantial and potentially increasing reliance on trade for its food availability is not seen as inherently problematic. The National Food Security Strategy, for example, rightly points out that 'being self-sufficient does not guarantee food security',²² as domestic harvest failure can get rid of that self-sufficiency in a matter of months. Indeed, as food production and trade became more globalised throughout the 20th and 21st centuries, global food security improved.²³ However, there are signs that the vulnerability of the global food system is increasing, which could have negative repercussions for the UK.

The global food system has grown increasingly interconnected since the beginning of the 20th century. Diets worldwide have experienced homogenisation – while in most places people now eat a greater variety of foods than in the past, human diets everywhere are increasingly similar.²⁴ This 'nutrition transition' generally involves an increased consumption of energy-dense foods (animal-sourced foods, plant oils and sugars) at the expense of traditional whole cereals, pulses and vegetables.²⁵ Mirroring the homogenisation of diets, global agricultural production has concentrated on

growing fewer crops of fewer varieties. In 2009, 50 crop commodities (from 94 crop species) accounted for 90% of calories, fat and weight consumed globally.²⁶ Crop genetic diversity has declined as a result – the UN Food and Agriculture Organization (FAO) estimated that over the 20th century, 75% of crop-species variety had been lost.²⁷

There are signs that the vulnerability of the global food system is increasing, which could have negative repercussions for the UK

The homogenisation in diets and food production has been enabled by rapidly increasing world trade in food and related products over the last century as well as by its increasing concentration in fewer hands.²⁸ The World Trade Organization's (WTO) Agreement on Agriculture, signed by WTO member states in 1995, liberalised global agricultural trade and spurred its growth.²⁹ The value of annual global food exports rose from \$13 billion in 1988 to \$820 billion in 2021.³⁰ More than a quarter of the food produced for human consumption is nowadays traded internationally³¹ and about 80% of the world's population lives in countries that are at least partially reliant on imported food.³² While most countries

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UK and Global Food Security in the Era Of 'Permacrisis'

participate in food trade in some capacity, just seven countries – Argentina, Australia, Brazil, Canada, New Zealand, Thailand and the US – account for 55% of food exports globally.³³ Concentration is also high and growing among companies active in the seed and agrochemicals trade, with just four companies controlling 62% of the world's pesticide market³⁴ and 70% of global trade in grain and oil seeds.³⁵ With increased trade, global food and agricultural-input supply chains have become longer, more complex and consequently more fragile.³⁶ The global food system has also become more financialised, which has encouraged large investments in commodity markets when prices are high, exacerbating global food-price volatility.³⁷

Another consequence of the increased global trade and concentration in food and agricultural inputs, in combination with high agricultural subsidies in many high-income countries, has been an increase in the number of net food-importing countries.³⁸ In fact, under the liberalised agricultural trade rules, many low- and middle-income countries (LMICs) have found themselves unable to compete with the subsidised agriculture from the Global North, leading to a reduction in their domestic food production, and an increase in food imports.³⁹ Just in the past decade, Africa's food imports have tripled in value;⁴⁰ in 2016–18, about 85% of the food consumed in Africa was imported.⁴¹ High reliance on food

imports combined with often low reserves of foreign currency have rendered net food-importing LMICs particularly vulnerable to disruptions in global food supply chains.

Throughout the 20th and beginning of the 21st century, the increasing interconnectedness of global agricultural and food systems was accompanied by improvements in global food security.⁴² Although, as mentioned earlier, countries' sufficient food availability does not automatically lead to satisfactory food access for all inhabitants, the two are positively correlated, and from 1950 to 2017, the world's proportion of food-insecure population declined spectacularly.⁴³ While about 65% of the global population was estimated to be undernourished (lacking access to sufficient daily calories) in 1950, the proportion dropped to 25% in 1970, to 15% in 2000 and to 8% in 2017.⁴⁴ However, since then, the global proportion of undernourished population has risen again, propelled among other factors by the increasing disruptions of the globalised food system.

Ongoing and Potential Disruptions to the Globalised Food System

The Covid-19 pandemic (2020–23) contributed significantly to the recent deterioration in global food security. While it did not majorly hinder global food

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44. *Ibid.*; World Bank, 'Prevalence of Undernourishment (% of Population)', <<https://data.worldbank.org/indicator/SN.ITK.DEFC.ZS>>, accessed 1 February 2024.

production, it notably weakened global food trade during its first months, exposing the relative fragility of the interconnected, financialised and highly concentrated global food system.⁴⁵ Subsequently, the pandemic-related public health measures and economic disturbances hindered financial and physical access to food in most countries,⁴⁶ including in the UK, where the inability to eat a healthy and nutritious diet rose from 3.2% in April 2020 to 16.3% just three months later.⁴⁷ The FAO estimated that 150 million additional people worldwide were food insecure in 2021 as compared to 2019 as a result of the pandemic.⁴⁸

The Russia–Ukraine war had a similar negative effect on global food security, brought about by the disruption of global agricultural trade.⁴⁹ Before the war, Russia and Ukraine were major grain and oil producers and exporters, together accounting for 73% of all sunflower oil, 34% wheat, and 17% maize traded globally.⁵⁰ Russia is also a major producer and exporter of oil, gas and chemical fertilisers. In the aftermath of Russia’s invasion of Ukraine in February 2022 and the associated imposition of sanctions on Russia and Belarus, global prices of food, energy and fertilisers skyrocketed. In March 2022, the FAO Food Price Index reached its highest level since its inception in 1990; the price of natural gas quadrupled and that of fertilisers tripled compared

to the prices in 2020,⁵¹ leading the UK to organise a Food Security Summit in November 2023 to address ‘Putin’s weaponisation of Ukraine grain’.⁵²

The rise in global prices of food and agricultural inputs affected local food prices everywhere but the most negatively affected were low-income countries⁵³ – most of them net food-importing and often dependent on international aid, to which the UK has traditionally been a major donor – with food inflation in January 2023 reaching almost 40%.⁵⁴ The food security situation in these countries was further exacerbated by the negative impact of the Russia–Ukraine war on the UN World Food Programme (WFP), the multilateral institution in charge of distributing food assistance, which used to source 40% of its wheat from Ukraine.⁵⁵ While global food, energy and fertiliser prices declined again by the end of 2023, high local food-price inflation has largely persisted, remaining at 30% in low-income countries in the last quarter of 2023.

Other conflicts, both ongoing and potential, might disrupt the global food system further. The number of state and non-state violent conflicts in the world has doubled since 2010.⁵⁶ The 21st century has also seen a marked increase in the nature and complexity of both internal conflicts and internationalised internal conflicts (for example, Russia’s complex involvement in Yemen’s civil war or China’s security

45. Jennifer Clapp and William G Moseley, ‘This Food Crisis is Different: COVID-19 and the Fragility of the Neoliberal Food Security Order’, *Journal of Peasant Studies* (Vol. 47, No. 7, October 2020), pp. 1393–417.
46. Christophe Béné et al., ‘Global Assessment of the Impacts of COVID-19 on Food Security’, *Global Food Security* (Vol. 31, No. 100575, December 2021).
47. Hartwig Pautz and Damian Dempsey, ‘Covid-19 and the Crisis of Food Insecurity in the UK’, *Contemporary Social Science* (Vol. 17, No. 5, May 2022), pp. 434–49.
48. FAO, IFAD, UNICEF, WFP and WHO, *The State of Food Security and Nutrition in the World 2022: Repurposing Food and Agricultural Policies to Make Healthy Diets More Affordable* (Rome: FAO, 2022), p. 13.
49. FAO, IFAD, UNICEF, WFP and WHO, *The State of Food Security and Nutrition in the World 2023: Urbanization, Agrifood Systems Transformation and Healthy Diets across the Rural–Urban Continuum* (Rome: FAO, 2023).
50. Kibrom A Abay et al., ‘The Russia-Ukraine War: Implications for Global and Regional Food Security and Potential Policy Responses’, *Global Food Security* (Vol. 36, No. 100675, March 2023).
51. Rob Vos et al., ‘Despite Improved Global Market Conditions, High Food Price Inflation Persists’, IFPRI, 4 December 2023, <[REDACTED]>, accessed 1 February 2024.
52. UK Prime Minister’s Office, ‘PM Announces Support to Alleviate Global Impact of Putin’s Weaponisation of Ukrainian Grain’, press release, 7 September 2023, <<https://www.gov.uk/government/news/pm-announces-support-to-alleviate-global-impact-of-putins-weaponisation-of-ukrainian-grain>>, accessed 1 February 2024.
53. Petra Hellegers, ‘Food Security Vulnerability due to Trade Dependencies on Russia and Ukraine’, *Food Security* (Vol. 14, July 2022), pp. 1503–10.
54. *Ibid.*
55. Mark A Green, ‘Forty Percent of the World Food Program’s Wheat Supplies Come from Ukraine’, Wilson Center, 2 June 2022, <[REDACTED]>, accessed 1 February 2024.
56. UCDP, ‘Uppsala Conflict Data Program’, Department of Peace and Conflict Research, [REDACTED], accessed 1 February 2024.

UK and Global Food Security in the Era Of 'Permacrisis'

operations in Mali and South Sudan). The strikes on ships in the Red Sea by Yemen's Houthis, related to the current intense Israel–Gaza conflict, are already disrupting trade routes⁵⁷ and might again fuel food-price inflation.⁵⁸ The eruption of new or the escalation of existing conflicts – for example, a potential Chinese invasion of Taiwan, regional spread of the Israel–Palestine war or of the civil war in Sudan – could lead to further disturbance of global markets and supply chains, dragging UK military assets further into conflict or peacekeeping operations.

In addition to conflicts, which undermine food security directly in affected countries and indirectly in unaffected countries through trade disruptions, climate change has also contributed to the recent deterioration in global food security.⁵⁹ Increasing temperatures, changing precipitation patterns, and adverse weather events driven by climate change have already reduced crop yields in many regions and their negative effects on food production are poised to intensify.⁶⁰ While some predictions see the UK as a net short-term winner from climate change, with land becoming more arable, this could be offset by damage to crops from severe weather and flooding.⁶¹

Disruptions in agricultural trade linked to climate change have not yet garnered as much attention as those due to conflicts, but this is likely to change in the future. A recent study highlighted that existing climate models have underestimated the likelihood

of concurrent harvest failures across the world as a result of a strongly meandering jet stream, which brings increased risk of extreme weather events including droughts, heatwaves and increased precipitation, all of which are detrimental to crop yields.⁶² While simultaneous harvest failures in global breadbaskets would likely not lead to a complete breakdown in global food trade, they would significantly disrupt it, and thus also place pressure on the UK's food availability.

Geopolitics, Yemen and Food Supply to the UK

Recent events in Yemen – the gateway to the Red Sea, the Suez Canal, and thus the movement of goods from East Africa and India to the UK – should act as a stark reminder of the intersection between geopolitics and trade. UK food imports are less vulnerable than some other commodities, such as clothes and shoes manufactured in Asia, but grain and palm oil, as well as luxuries including coconut milk, chocolate, wine, spices and tropical fruits all make the journey north through the Suez Canal. Costs of all these are set to rise to compensate for the longer journey around South and Western Africa,⁶³ with 'inevitable' food inflation forecast.⁶⁴ The disruption also highlights the vulnerability of world trade to other choke points, such as the Strait of Gibraltar, Panama Canal and Strait of Malacca.⁶⁵

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57. Alessandro Ford, 'Spoilage Alert: Red Sea Crisis Hits Europe's Fresh Food Trade', *Politico*, 26 January 2024, <[REDACTED]>, accessed 1 February 2024.
 58. VOA, 'Houthi Attacks on Red Sea Shipping Leads to Economic Logjams', 28 January 2024, <[REDACTED]>, accessed 1 February 2024.
 59. Andrea Gomez-Zavaglia, Juan Carlos Mejuto and Jesus Simal-Gandara, 'Mitigation of Emerging Implications of Climate Change on Food Production Systems', *Food Research International* (Vol. 134, Article 109256, August 2020).
 60. Robyn Anderson, Philipp E Bayer and David Edwards, 'Climate Change and the Need for Agricultural Adaptation', *Current Opinion in Plant Biology* (Vol. 56, August 2020), pp. 197–202.
 61. James Rising et al., 'What Will Climate Change Cost the UK? A Study of Climate Risks, Impacts and Mitigation for the Net-Zero Transition', Grantham Research Institute on Climate Change and the Environment, London School of Economics and Political Science, 2022, <[REDACTED]>, accessed 1 February 2024.
 62. Kai Kornhuber, 'Risks of Synchronized Low Yields Are Underestimated in Climate and Crop Model Projections', *Nature Communications* (Vol. 14, Article 3528, July 2023).
 63. Cahal Milmo, Arj Singh and Sam Rucker, 'UK Faces Price Rises on Food, TVs and Trainers due to Red Sea Shipping Crisis', *iNews*, 19 December 2023, <[REDACTED]>, accessed 1 February 2024.
 64. Ian Quinn, 'Red Sea Shipping Attacks Will Lead to 'Inevitable' Food Inflation', *The Grocer*, 4 January 2024, <<https://www.thegrocer.co.uk/supply-chain/red-sea-shipping-attacks-will-lead-to-inevitable-food-inflation/686760.article>>, accessed 1 February 2024.
 65. Sarah Schiffling and Matthew Tickle, 'Red Sea Crisis: Suez Canal is Not the Only "Choke Point" that Threatens to Disrupt Global Supply Chains', *The Conversation*, 15 January 2024, <[REDACTED]>, accessed 1 February 2024.

For most of the 21st century, Yemen has been embroiled in an internal civil war that has dragged in Saudi Arabia, the US and the UK, while Russia has tried to maintain diplomatic relations with both the UN-recognised government and the Houthi rebel insurgents in an attempt to protect Russian interests in the region, including access to and control of trade routes. This has been severely tested by the Russia–Ukraine conflict’s impact on Yemen’s already dire food insecurity and will be tested further by the current conflict – particularly as it pulls Western allies further in.

Yemen itself is one of the world’s most food insecure countries. The long running internal crisis means that although the country should have sufficient arable land to feed its population, in-country farming production of previous key exports, such as coffee, have been decimated. Yemen is highly dependent on imported food, spending 15% of its GDP on food imports, and is thus extremely vulnerable to international price fluctuations and increases. Undernourishment in the country is high and water supplies are often contaminated: outbreaks of infectious waterborne disease including cholera are rife. More than 50% of Yemeni households are dependent on some type of food assistance, much of which is sourced from other MENA countries, including Egypt, the UAE and Turkey, which have traditionally been highly dependent on Russian/Ukrainian imports.⁶⁶ Such allegiances can be consolidated or fractured through food aid as soft power, but the influence of food supply on geopolitics beyond the immediate region, and on UK policy specifically, must not be underestimated.

Responses to Global Food System Disruptions

Increased food prices and deficient food access, resulting from disruptions to the global food system, have far-reaching consequences not only within the countries affected but also across borders. Food insecurity often provides a window of opportunity for populist leaders and the fossil fuel industry offering immediate, but environmentally unsustainable, ‘quick fixes’, which might damage transitions towards a greener economy. There is already evidence⁶⁷ that the UK far-right has been emboldened by populist opposition to green politics, which is pushing the government to backtrack on key net-zero policies that would help protect future food production. Food availability and affordability can in general be an important factor in popular government support, while steep food-price increases often lead to protests and riots.⁶⁸ For example, the ‘Arab Spring’ is believed to have been partly triggered by food-price increases during the 2008–09 food crisis.⁶⁹ Finally, severe food insecurity fuels distress migration, increasing numbers of internally displaced people and refugees,⁷⁰ which might trigger new or further exacerbate existing conflicts. The experience of the first few weeks of the Covid-19 pandemic, with empty supermarket shelves and food hoarding, is a potent reminder of how quickly food shortages can lead to civil disruption in the UK too.

To avoid these ramifications, net food-exporting countries often react to crises by banning or restricting their food exports. During the 2008–09 global food crisis, 32 countries placed restrictions on some food exports, usually on staple grains such as wheat, rice, maize or soybeans.⁷¹ Similarly, in the

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66. Jennifer Cole et al., ‘Conflict, Collapse and Covid-19: Lessons from Yemen on Elevated Disease Risk in Regions Under Stress’, *RUSI Journal* (Vol. 166, No. 3, 2021), pp. 10–19.
 67. Garrett L Grainger, ‘Sunak’s Rhetoric on Climate Action May Embolden Britain’s Far Right’, 5 October 2023, LSE, [redacted] >, accessed 1 February 2024.
 68. Davide Natalini, Giangiacomo Bravo and Aled Wynne Jones, ‘Global Food Security and Food Riots – An Agent-Based Modelling Approach’, *Food Security* (Vol. 11, July 2017), pp. 1153–73.
 69. Giulia Soffiantini, ‘Food Insecurity and Political Instability during the Arab Spring’, *Global Food Security* (Vol. 26, Article 100400, September 2020).
 70. Giovanni Anania, ‘Agricultural Export Restrictions and the WTO: What Options do Policy-Makers have for Promoting Food Security?’, International Centre for Trade and Sustainable Development, Issue Paper No. 50, November 2013.
 71. *Ibid.*

UK and Global Food Security in the Era Of 'Permacrisis'

first months of the Covid-19 pandemic, 21 countries put in place export restrictions, affecting 8% of all calories traded globally,⁷² and following Russia's invasion of Ukraine in February 2022, 74 countries restricted or banned exports of some foodstuffs.⁷³ The export restrictions often, although not always, help reduce domestic food prices in the countries that impose them,⁷⁴ but simultaneously, by restricting global food supply, they contribute to higher food prices elsewhere. The export restrictions imposed because of the Ukraine–Russia war were estimated to have increased the global price of rice by 12% and of wheat by 9%.⁷⁵ The impacts of food-export restrictions in the UK could be mitigated with forward-planning and stockpiling, but this does not sit easily with the prevalent 'just in time' supply chain culture.

However, some net food-importing countries have already reacted to the increasing disruptions of the global food system, either by increasing domestic food production or by diversifying trade partners. China has attempted to do both. Its food self-sufficiency has steadily decreased, from 94% in 2000 to 66% in 2020, but driven by fears that global food trade could become weaponised, since 2023, China has attempted to increase local food

production again by expanding its agricultural land.⁷⁶ At the same time, China has decreased its food-import reliance on the US, reducing the share of US farm products in food imports from 26% in 2012 to 18% in 2022,⁷⁷ while increasing imports from Brazil. Along with India and South Africa, Brazil is an increasingly important global food exporter, which is somewhat diversifying the global food trade system;⁷⁸ from just three major agriculture trade communities before 2010, there are now five, with Brazil–South Asia (as well as South Africa) becoming a serious global contender.⁷⁹ Even countries in the Middle East with climatic conditions inimical to agriculture, such as Saudi Arabia and the UAE, have begun trying to reduce their food-import dependency by investing in vertical farming technology,⁸⁰ drought-resistant biotechnology⁸¹ and in purchasing farmland overseas, for example in the US.⁸² Vertical farming is growing in popularity also in the UK, due to its lower space and water requirements, climate-controlled conditions, and year-round production.⁸³

Disruptions to established trade routes can sometimes inspire positive innovation. The decline in UK food imports during the Second World War led to the introduction of food rationing, higher consumption of fruits and vegetables, and lower

72. Joseph Glauber, David Laborde and Abdullah Mamun, 'From Bad to Worse: How Russia-Ukraine War-Related Export Restrictions Exacerbate Global Food Insecurity', IFPRI, 13 April 2022, [redacted], accessed 1 February 2024.
73. Alvaro Espitia, Nadia Rocha and Michelle Ruta, 'How Export Restrictions Are Impacting Global Food Prices', World Bank Blogs, 6 July 2022, <[redacted]>, accessed 1 February 2024.
74. Sonia Akter, 'The Effects of Food Export Restrictions on the Domestic Economy of Exporting Countries: A Review', *Global Food Security* (Vol. 35, Article 100657, December 2022).
75. Espitia, Rocha and Ruta, 'How Export Restrictions Are Impacting Global Food Prices'.
76. Lin Ganfeng, 'To Increase Food Security, China Expands Farmland', VOA, 23 August 2023, <<https://www.voanews.com/a/to-increase-food-security-china-expands-farmland/7237866.html>>, accessed 1 February 2024.
77. Chad P Bown and Yilin Wang, 'Five Years into the Trade War, China Continues its Slow Decoupling from US Exports', Peterson Institute for International Economics, 16 March 2023, <[redacted]>, accessed 1 February 2024.
78. See Josh Lipsky and Mrugank Bhusari, 'Brazil Aims to Advance Its Bid for Leadership of the Global South through Food Security', Atlantic Council, 14 February 2024, <[redacted]> />, accessed 27 February 2024.
79. See Xian Wang et al., 'Trade for Food Security: The Stability of Global Agricultural Trade Networks', *Foods*, (Vol. 12, No. 2, 2023).
80. Meshal J Abdullah, Zhengyang Zhang and Kazuyo Matsubae, 'Potential for Food Self-Sufficiency Improvements through Indoor and Vertical Farming in the Gulf Cooperation Council: Challenges and Opportunities from the Case of Kuwait', *Sustainability* (Vol. 13, No. 22, Article 12553, November 2021).
81. Seema R Bajaj et al., 'First Report of European Truffle Ectomycorrhiza in the Semi-Arid Climate of Saudi Arabia', *3 Biotech* (Vol. 11, Article 24, January 2021).
82. Ian James, 'Saudi Firm that Grows Hay in California and Arizona To Lose Farm Leases over Water Issue', *Los Angeles Times*, 5 October 2023.
83. Evengate Financial Services, 'Vertical Farming UK – the Future or a Phase?', [redacted] />, accessed 1 February 2024.

consumption of meat and sugar. Many people consequently enjoyed healthier diets, which contributed to lower infant mortality and a higher average age of death from natural causes.⁸⁴ Similarly, following the collapse of trade with the Soviet Union in 1989, Cuba managed to revitalise its domestic agriculture and strengthen food security in an environmentally sustainable way. Immediately following the collapse, Cuban food availability drastically decreased, as the country lacked foreign currency to import food or agrochemicals from other countries. However, Cuba managed to rapidly increase domestic food production through the expansion of agricultural land, including the establishment of thousands of urban gardens, and development of a successful system of organic farming.⁸⁵ Within the next decade, food insecurity in Cuba declined to levels below those in 1989.⁸⁶

Some net food-importing countries have already reacted to the increasing disruptions of the global food system, either by increasing domestic food production or by diversifying trade partners

However, such success stories are uncommon. Sri Lanka recently attempted to emulate Cuba's transition to organic agriculture but with disastrous results, largely due to poor management of the transition. Faced with a shortage of foreign currency during an economic crisis, the Sri Lankan government banned virtually overnight the imports of chemical fertilisers.⁸⁷ The ban, combined with fuel and water shortages, led to widespread harvest failures, and was withdrawn six months later amid growing food insecurity. Many other LMICs also

currently face severe food insecurity but cannot easily raise domestic food production, whether due to ongoing conflicts, adverse climate conditions, foreign currency shortages, or the inability to compete with cheaper food imports from countries with subsidised agricultural production in a liberalised agricultural trade regime. Moreover, due to the escalating 'permacrisis', global availability of humanitarian assistance has decreased, with the WFP presently facing a 60% funding shortfall.⁸⁸

Increasing global instability resulting from worsening food insecurity has direct implications for UK national security – but do these issues carry any relevance for UK food security as well? As noted, the UK is not self-sufficient in food production, and this might decline further in the future. However, the country has ensured sufficient domestic food availability through food imports for decades and, post-Brexit, it has diversified its portfolio of agricultural trading partners through new free-trade deals, which could enhance the resilience of its food supply chains. Nevertheless, as a net food importer, the UK is arguably more exposed to global agricultural trade disruptions than net food-exporting countries. That food-price inflation in the UK in 2023 was higher than in all large net food-exporting European countries provides some evidence for this claim.⁸⁹ The more restricted post-Brexit trade between the UK and EU countries (many of them large food producers) makes UK food imports yet more vulnerable during crises, as illustrated by the shortage of fresh vegetables in the UK in spring 2023.⁹⁰

Recommendations

UK food security and the global food system are connected. The UK is not self-sufficient in food production, but as the global food production and trade system has grown more interlinked

84. Michael Denham, 'As We Once Were: Wartime Rationing', British Geriatrics Society, 14 November 2015, [redacted], accessed 1 February 2024.
85. Sinan Koont, 'Food Security in Cuba', *Monthly Review* (Vol. 55, No. 8, January 2004), pp. 11–21.
86. Food and Agriculture Organization, [redacted], accessed 1 February 2024.
87. Nick Young, 'Sri Lanka's Fertiliser Ban and Why New Zealand Can Phase Out Synthetic Nitrogen Fertiliser', Greenpeace, 14 December 2022, <<https://www.greenpeace.org/aotearoa/story/on-sri-lankas-fertiliser-ban/>>, accessed 1 February 2024.
88. New Humanitarian, 'What WFP Cuts Mean for People in Hunger Crises around the World', 13 December 2023, [redacted], accessed 1 February 2024.
89. Anthony Reuben, 'Is Food Inflation Higher in Europe than in the UK?', *BBC News*, 20 June 2023.
90. Sam Jones, 'Spanish Growers Say Weather, Rising Costs and Brexit Caused UK Salad Shortages', *The Guardian*, 23 February 2023.

UK and Global Food Security in the Era Of 'Permacrisis'

throughout the past century, this has correlated with improvements in global and domestic food security. Nevertheless, the global food system has experienced more disruptions in recent years and is poised to see more such disruptions in the future, related to conflicts, climate change, or most likely both.

A global food system in which production is more widely distributed could be more resilient, with fewer single points of failure, more self-sufficiency, and more stable prices. One step towards this would be renegotiation of the WTO Agreement on Agriculture to allow net food-importing LMICs to increase their domestic food production, which would reduce the concentration in global food production and thus reduce the likelihood of trade disruption. Greater global investment in agricultural research could improve the resilience of food production in the face of accelerating climate change. International agreements guaranteeing the movement of food trade even amid conflicts, along the lines of the Russia–Ukraine Black Sea grain deal, could help ameliorate food trade disruptions triggered by geopolitical instability. These actions would all strengthen global food security and, along with it, the food security of the UK.

To specifically bolster UK food security, however, it may be necessary to ensure both the diversification of food and other trade routes, and uninhibited trading relationships, particularly with regional

neighbours. A more radical approach would be to also consider specifying a minimum level of food that has to be produced domestically, with subsidies and support to encourage agricultural production built into future resilience and security budgets.⁹¹

Increased domestic food production could be attained through investment in urban gardening as well as changes to agricultural policy to make crop and livestock farming more attractive to UK farmers. The latter could bolster national security and resilience from a number of directions: promoting jobs and careers in the rural economy; encouraging research into environmental protection (particularly to avoid further degradation of soil) and sustainable farming methods; as well as ensuring UK food security in the near- and longer-term future. ■

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91. UK farmers have been demanding more financial support and in February 2024 the UK Government provided them with some additional funding to help increase UK food production. See Malcolm Prior and Jonah Fisher, 'PM Announces Support for Farming amid Industry "Crisis"', *BBC News*, 21 February 2024.

April 2019



How Britain became one of the most nature-depleted countries in the world

Can a country truly claim to be a global climate leader while its own nature crumbles? Paul de Zylva, Friends of the Earth's Senior Nature Analyst, investigates the gap between the UK's green rhetoric and the grim reality of its biodiversity collapse. Britain's natural wealth is in ruins, and as the 2030 deadline approaches, here's what we need to do to reverse the damage.

Contents

1. [The evidence shows decades of decline](#)
2. [One cause: Policy failure](#)
3. [Second cause: Legal backpedalling](#)
4. [Our economy and overseas territories](#)
5. [Turning the tide before 2030 is possible](#)

The United Kingdom prides itself on being a green and pleasant land, a nation synonymous with rolling countryside, abundant wildlife, and environmental stewardship. Yet beneath this postcard-perfect image lies a startling reality: the UK is now one of the world's most nature-depleted countries.

According to the Natural History Museum's Biodiversity Intactness Index, **the UK ranks 189th out of 240 countries** and territories for how intact its nature and biodiversity remain. Just 53% of the UK's original nature is left intact - far below the Index's safe limit of 90% and the global average of 79%. Each of the four UK nations sits near the bottom of this ranking, with England at 47%, Northern Ireland at 50%, Wales at 51%, and Scotland at 56%.

This is a crisis.

As the 2030 deadline approaches (the UK government's legally binding statutory targets under the Environment Act 2021), the question is no longer whether the UK is failing nature: it's whether there's still time to turn the tide.

The evidence shows decades of decline

The data tells a story that no amount of green rhetoric can obscure. Across nearly two decades of monitoring, the UK's biodiversity has shown stubborn stagnation at best and outright decline at worst.

The UK government tracks approximately 50 different biodiversity indicators. The 2025 results paint a troubling picture.

Indicators still heading in the wrong direction include:

- Condition of important habitats
- Abundance and distribution of wild species
- Size of fish in the North Sea
- Status of pollinating insects (bees, hoverflies, moths)
- Bird numbers on farmland and in woodlands

Critically, the December 2025 indicators confirmed what conservationists feared: "...more indicators are deteriorating or showing no change than improving, over both the long and short term." Even more alarming: "No indicators which are deteriorating in the long term, are showing improvement in the short term."

The State of Nature Reports: The timeline of decline

Four major State of Nature reports have documented the accelerating crisis:



The Biodiversity Intactness Index: Where the UK stands globally

The Natural History Museum's Biodiversity Intactness Index (BII) tracks the percentage of original species remaining in habitats and nations. The UK's position is stark:



For comparison, our French neighbours are doing considerably better than us in the top 2% of countries in the lofty 5th position, New Zealand – another small island nation, are also near the top at 27th globally. Even Japan who have a higher population density to us, rank 55th with 78% of their biodiversity still intact.

Professor Andy Purkis of the Natural History Museum noted: "*The UK is where the industrial revolution was born, and it changed the landscape forever. We have led the world in degrading the natural environment... Britain has lost more of its natural biodiversity than almost anywhere else in western Europe, most of all the G7 nations.*"

Positive signs (few, but notable)

The 2025 indicators did show some improvements:

- Record improvements in bat species abundance
- Percentage of fish stocks being harvested sustainably
- Amount of land being managed in environment-friendly ways

While these gains are overshadowed by the broader trend of deterioration across sea, farmland, and woodland ecosystems, perhaps there are lessons to be learned from these positive signs – but more on that later.

What we don't know

Perhaps most worryingly, **over 30% of the UK's biodiversity indicators weren't even assessed in the 2025 report**. This means no measure exists for:

- Status of bees and other pollinating insects
- Condition of protected Areas/Sites of Special Scientific Interest (A/SSSI)
- Pressure from invasive species
- Extent of green and blue spaces in urban areas
- How public subsidies might be undermining environmental aims

This data gap makes it impossible to know the full extent of the crisis or to measure whether interventions are working.

The bottom line

From 2008 to 2025, the message has remained consistent: England's natural environment has suffered serious losses. Wildlife-rich sites form a "*small isolated and fragmented resource*." Protected areas are "*generally too small and too isolated*." And despite the 25 Year Environment Plan's bold promise to "*leave the environment in a better state than we found it*," the indicators show stubborn or stalling progress at best.

The UK's biodiversity is in decline, the rate is accelerating, and the policy response has been insufficient.

One cause: Policy failure

The UK government has a long history of setting ambitious nature targets, only to miss them. This pattern of failure has created a cycle of "dither and delay" that has pushed nature to the brink.

Missed deadlines

The UK's failure to protect nature is not a recent phenomenon; it is a decades-long trend of unmet commitments:

- **2010 Targets:** The UK helped negotiate the **2010 Nagoya Protocol**, aiming to stop and reverse harm to nature by 2010. The government admitted it would only meet 5 of the 20 targets. The RSPB described this failure as "**a lost decade for nature.**"
- **2020 Targets:** History repeated itself. The UK missed its 2020 targets for reversing nature's decline. In the most crucial areas: controlling pollution, restoring ecosystems, and the status of wild species, the UK was "**nowhere near where it should have been,**" despite its aspiration to lead the world.
- **2030 Targets:** Now, with the statutory deadline to halt nature's decline looming, the UK looks set to miss these targets as well. As I stated above the 2025 Biodiversity Indicators confirm that "*no indicators which are deteriorating in the long term, are showing improvement in the short term,*" suggesting the negative trend has not been reversed.

The 25-Year Environment Plan: Bold words, little action

In 2018, recognising the dire situation, the UK government launched its 25 Year Environment Plan with the bold aim of "*improving the environment within a generation and leaving it in a better state than we found it.*"

These were bold words indeed, but of little use until the government takes proper action. Given the decades of failure by successive governments in meeting previous promises, the need for immediate action has never been greater. The plan's promise remains unfulfilled, with the 2025 indicators showing that "*more indicators are deteriorating or showing no change than improving.*"

The pattern is clear: **governments set targets, fail to meet them, and then attempt to weaken the laws designed to ensure those targets are met.** As the 2030 deadline approaches, the UK risks another "lost decade" for nature, unless the cycle of broken promises is itself finally broken.

Second cause: Legal backpedalling

As an EU member, the UK had signed up to EU standards and laws that have been instrumental in protecting many wild species and habitats, as well as raising standards for drinking water, clean beaches and bathing water, and air quality.

Signs were positive when the government pledged that when we left the EU, it would retain and abide by the standards the UK had helped design. During and since the 2016 Brexit referendum, government ministers and MPs have made countless pledges to retain existing EU standards - and even to exceed them - in a new post-Brexit global leadership role for the UK on nature, climate and other aims.

The promise: Environment Act 2021

On leaving the EU, the UK government enshrined existing EU standards and the UK's future ambitions in a new "landmark" Environment Act. It became law on 10 November 2021, with the government's press release proclaiming: *"World-leading Environment Act becomes law."*

The then Environment Secretary George Eustice repeated this claim:

"The Environment Act will deliver the most ambitious environmental programme of any country on earth. It will halt the decline of species by 2030, clean up our air and protect the health of our rivers, reform the way in which we deal with waste and tackle deforestation overseas. We are setting an example for the rest of the world to follow."

Prime Minister Boris Johnson also pledged to **protect 30% of our land and seas by 2030**, as part of the global Leader's Pledge for Nature. Johnson said: *"We cannot afford dither and delay because biodiversity loss is happening today, and it is happening at a frightening rate."*

The Environment Act 2021's legally binding commitment is to **reverse the decline in the abundance of wild species by the end of 2030**, and to start increasing species populations into the 2040s and 2050s.

The contradiction: Three attempts to weaken nature laws

Yet even with the ink still drying, senior ministers were trying to weaken or scrap two vital nature laws: the **1979 Birds Directive** and the **1992 Habitats Directive**. These are the legal backstops that protect the most precious wildlife species, natural habitats and landscapes across the UK and Europe.

How do UK governments expect to meet their own commitments if at the same time they're trying to gut the most important laws? Again, the UK has form on this. Since 2010 ministers have tried to disable or scrap all-important nature laws – not once, nor twice, but three times:

- **2011:** Chancellor George Osborne claimed nature laws were blocking progress and should be weakened. A formal government review concluded his claims were "not correct."
- **2016:** A "fitness check" review found that **nature laws were not blocking economic development**, contradicting government claims.

- **2023:** Ministers are once again looking to meddle with the very laws needed to meet their own pledges.

The public pushback

Friends of the Earth, WWF, Birdlife and over 100 other UK conservation organisations publicised the threat to fundamental laws protecting vulnerable habitats and endangered species. Our Nature Alert campaign saw **100,000 people from the UK** join with people across EU member states to give a resounding "no" to any attempts to dilute or scrap the nature laws.

Half a million people told the official consultation that they simply wanted their governments to get on with properly observing and implementing the nature laws - not trying to get round them.

So, it's baffling that UK ministers are once again looking to meddle with laws they simply need to start observing. They don't seem to understand that to meet their own pledges to protect and restore nature across the UK in the timescale needed (including to meet international obligations such as on migratory species and internationally important habitats) they don't have time to scrap EU standards and dabble with new ones, which could take years to become enacted.

Instead, ministers must step up and start observing existing nature laws, and their own pledges, instead of peddling falsehoods that such laws are either a restraint on the UK or that they can be bettered.

Our economy and overseas territories

The UK government cannot claim ignorance of the true cost of putting nature out of business. The Treasury, in particular, knows this well, having commissioned the first-ever independent review on the subject.

The economics of biodiversity

In the 2021 Economics of Biodiversity review, leading economist Partha Dasgupta advised ministers:

*"Continuing down our current path – where **our demands on Nature far exceed its capacity to supply – presents extreme risks and uncertainty for our economies**. Sustainable economic growth and development requires us to take a different path, where our engagements with Nature are not only sustainable, but also enhance our collective wealth and well-being and that of our descendants."*

Attempts by any UK government to scrap our nature laws are deeply misguided, short-sighted, and even reckless when there is no evidence that such laws block or thwart the UK's economic aims. The cost of inaction is not just ecological; it is financial.

The UK's nature hotspots: Overseas Territories

And what about nature outside Britain? The UK's responsibility extends far beyond its shores. **Over 90% of the UK's biodiversity is found in its Overseas Territories** (UKOTs), located in some of the world's biodiversity hotspots, such as the Caribbean, Malta, and Cyprus.

These territories are home to:

- Many **endemic species** and undisturbed habitats of international significance.
- **33 Red List bird species** - more than the whole of continental Europe.
- A combined **32,000 species**, of which **at least 517 are globally threatened**.
- Even excluding the 60,000 species yet to be properly recorded, this dwarfs the 194 globally threatened species found in mainland UK itself.

The UKOTs also span large stretches of the South Atlantic, Indian, and South Pacific oceans. Consequently, the UK effectively manages the **fifth largest area of ocean in the world** and potentially the most biodiverse. An important step has been the creation of vast marine protection areas based around Pitcairn in the Pacific and Ascension in the Atlantic.

Threats and failures

However, biodiversity in the UKOTs faces immediate and significant threats, particularly from:

- Invasive species
- Under-regulated development

- Over-exploitation of natural resources
- Climate change

A 2014 inquiry by UK MPs found that the UK government was **failing to adequately protect the globally significant nature** in its Overseas Territories.

UK failing home and away

The UK likes to say it is leading the world in halting nature's decline, but its own official assessment is clear: the UK is failing nature both at home and away.

In February 2026, the UK's own assessment of its nature and biodiversity targets (domestically and overseas) shows the UK mostly going backward as the 2030 deadline looms. The UK 7th National Report states:

- *"Biodiversity decline persists"*
- *"The overall trend in status for many wild species in the UK is declining or not improving"*
- *"Most (UK) ecosystems remain threatened, and only 24% of land is covered by natural ecosystems"*
- The 'Red List of Ecosystems' shows that, of those assessed, most are classified as *"critically endangered"*, *"endangered"* or *"vulnerable"*.

The high point of the assessment was, at best, neutral: *"Numerous restoration actions are currently underway and progress is being made, the effectiveness of these efforts and their contribution to restoring the integrity, connectivity and resilience of all ecosystems and increasing the area of natural ecosystems remains uncertain."*

Overall, the report concludes: *"...work is required to address key gaps in halting species decline, improving ecosystem resilience, and safeguarding genetic diversity in the UK."*

Turning the tide before 2030 is possible

The evidence is overwhelming, the failures are documented, and the risks are clear. But despair is not an option. The UK stands at a crossroads: continue down the path of "dither and delay" and face irreversible loss or seize the remaining window of opportunity to restore nature at pace and scale.

The path forward is not a mystery; it's a matter of political will and decisive action. Here is what must happen to turn the tide before the 2030 deadline:

1. Stop meddling, start enforcing

The most immediate step is to stop trying to weaken the laws that already exist. The 1979 Birds Directive and the 1992 Habitats Directive are not obstacles; they are the bedrock of nature protection.

- **Action:** Halt all attempts to scrap or dilute these vital laws.
- **Action:** Fully implement the Environment Act 2021 as written, ensuring the legally binding 2030 targets are met, not just aspirational goals.
- **Why:** As the 2011 and 2016 reviews proved, these laws do not block economic progress, they enable sustainable growth.

2. Fill the data gaps

We cannot manage what we do not measure. The fact that over 30% of biodiversity indicators remain unassessed is a big blind spot.

- **Action:** Immediately fund and launch assessments for the missing indicators: pollinator status, protected site conditions, invasive species pressure, and urban green/blue spaces.
- **Action:** Ensure public subsidies are audited to confirm they are not undermining environmental aims.
- **Why:** Without this data, we are flying blind. Accurate monitoring is the only way to know if restoration efforts are working and to hold government accountable.

3. Fund restoration at scale

The Dasgupta Review made it clear: sustainable economic growth depends on a healthy natural world.

- **Action:** Redirect public subsidies toward nature-positive outcomes, rewarding farmers and landowners who restore habitats rather than degrade them.
- **Action:** Invest in large-scale habitat connectivity to fix the "fragmented resource" problem identified in 2008. Protected areas must be linked to allow species to move and thrive.
- **Action:** Protect the Overseas Territories with the same vigour as the mainland, addressing invasive species and over-exploitation to safeguard 90% of the UK's biodiversity.

A new narrative: Hope in action

There is reason to be hopeful. Nature is resilient. Where we have given it space, it has bounced back.

- Remember the **record improvements in bat species** and the **sustainable fishing stocks** highlighted in the 2025 indicators. These prove that recovery is possible when we act.
- The **power of public opinion shouldn't be underestimated** - the 100,000+ people who signed the Nature Alert campaign and the half a million who demanded proper implementation show that the public will back politicians who will finally implement change.
- The UK has the chance to lead the world not by talking, but by doing. By fixing its own backyard and protecting its Overseas Territories, it can **set a global example**.

The choice is ours

The 2030 deadline is a finish line we are currently running away from. But it isn't too late.

- We can choose to listen to the scientists and the data.
- We can choose to enforce the laws we have already passed.
- We can choose to invest in our natural wealth rather than deplete it.
- We can choose to see nature protection as the basis for sustainable growth rather than as a mass of red tape to be jettisoned.

Our leaders must stop pushing nature to the edge and start restoring our degraded environment at pace and at scale.

The UK has the tools, the knowledge, and the public support. What it needs now is the courage to use them. The future of Britain's nature - and the economic and social well-being of its people - depends on the choices made today.



Article

Are Rurality, Area Deprivation, Access to Outside Space, and Green Space Associated with Mental Health during the COVID-19 Pandemic? A Cross Sectional Study (CHARIS-E)



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Abstract: The study investigated if rurality, area deprivation, access to outside space (Study 1), and frequency of visiting and duration in green space (Study 2) are associated with mental health during the COVID-19 pandemic and examined if individual demographics (age, gender, COVID-19 shielding status) and illness beliefs have a direct association with mental health during the COVID-19 pandemic. A serial, weekly, nationally representative, cross-sectional, observational study of randomly selected adults was conducted in Scotland during June and July 2020. If available, validated instruments were used to measure psychological distress, individual demographics, illness beliefs, and the following characteristics: Rurality, area deprivation, access to residential outside space, frequency of visiting, and duration in green space. Simple linear regressions followed by examination of moderation effect were conducted. There were 2969 participants in Study 1, of which 1765 (59.6%) were female, 349 (11.9%) were in the shielding category, and the median age was 54 years. There were 502 participants in Study 2, of which 295 (58.60%) were female, 58 (11.6%) were in shielding category, and the median age was 53 years. Direct effects showed that psychological distress was worse if participants reported the following: Urban, in a deprived area, no access to or sharing residential outside space, fewer visits to green space (environment), younger, female, in the shielding category (demographics), worse illness (COVID-19) representations, and greater threat perception (illness beliefs). Moderation analyses showed that environmental factors amplified the direct effects of the individual factors on psychological distress. This study offers pointers for public health and for environmental planning, design, and management, including housing design and public open space provision and regulation.

Keywords: COVID-19; environment; place; rural; urban; area deprivation; green space; mental health; pandemic

1. Introduction

During the COVID-19 pandemic, governments have introduced legislation and guidelines to prevent the spread of the coronavirus, which have restricted movement and social interactions and have frequently confined people within small geographical areas. Prior to the pandemic, various indices of the environment have been associated with mental health [1,2]. Hence, one might expect relationships between the environment and mental health to be particularly salient during the current pandemic. Environment is a concept with several definitions and dimensions [3], including rurality, area deprivation, residential

outside space, and green and natural environments. However, there are other factors related to mental health that may also be important in the context of the pandemic.

There is ample evidence that individual demographic factors are related to mental health. During the COVID-19 pandemic, for instance, women have consistently reported more anxiety and depression than men and young people have reported greater anxiety than older people [4–6]. In addition, illness beliefs have been associated with mental health during the current pandemic. A growing number of studies have used the concept of risk perception to understand the effects of the pandemic on mental health [6–10]. These studies have shown that different dimensions of the construct risk perception are associated with mental health. Other beliefs, such as beliefs about the illness COVID-19, have also been found to be associated with mental health during the pandemic [6], which is consistent with the substantial body of empirical work based on the Common Sense Self-Regulation Model (CS-SRM) [11] and with the CS-SRM as theoretically conceived [12]. CS-SRM proposes that people construct representations of a health threat, which help them make sense of their experiences and provides a basis for their own coping responses. Beliefs about illness are central to the model and incorporate five key components: Beliefs about the nature (identity), time-course (timeline), personal impact (consequences), causal factors (cause), and feasibility of control or cure (control/cure) of the illness

It is conceivable that different environments would amplify the effects of these individual demographic factors and illness beliefs on mental health during the COVID-19 pandemic. Systematic reviews have concluded that characteristics of the environment are associated with depressive mood and psychological distress [13,14]. Associations have been found between area deprivation and death by suicide, anxiety, and depression [15–17], but other studies have suggested that this can be explained by individual (e.g., age, gender, marital status) rather than environmental factors [17,18]. Systematic reviews and meta-analyses have highlighted differences between people living in rural and urban areas for depression and psychiatric disorders, with some studies showing that mental health is worse in urban compared to rural areas and other studies suggesting the opposite [19,20]. The availability of green space might be one important factor in explaining urban-rural differences in mental health [21] and the link between area deprivation and stress [22,23]. Several theoretical hypotheses that have been proposed to explain how exposure to the natural environment may impact mental wellbeing [24,25]. These include a suggestion that humans have an innate evolutionary connection to the natural environment such that any exposure to it will reduce levels of stress and improve mood (“biophilia hypothesis”) [26,27] and a conceptualization of natural places as ‘restorative environments’ that mediate the negative effects of stress (“psycho-evolutionary stress reduction theory”) [28].

A pre-pandemic study found that people spending ≥ 120 min per week in natural environments reported good health and well-being compared to people with no contact with nature and that positive associations peaked at between 200–300 min per week with no further gain [29]. The extent to which people use green space during the pandemic may have changed due to the temporary closure of parks and golf courses and restrictions on travel and opportunities to meet with others. Nonetheless, uncertainties about the influence of environmental factors on mental health remain [30]. For example, a systematic review of urban green space exposure reported mixed results with associations found between green space and mood but not for depression and stress [31].

Taken together, the above body of work calls for the examination of the impact of environment both directly and in moderating individual demographic and psychological factors that are associated with mental health during pandemics. The research may be useful to health care planners by highlighting which modifiable environmental factors could be used to improve mental health during pandemics. To date, there are few studies about the relationship between environment and mental health in the context of pandemics, yet home lockdown and restrictions on travel during the COVID-19 pandemic have heightened the importance of the environment as an aspect of people’s lives. This explains the focus of our paper.

An aim of the COVID-19 Health and Adherence Research in Scotland (CHARIS) project [32] was to investigate mental health in the adult population during the COVID-19 pandemic and to explain variations in mental health. Hence, two CHARIS studies examined the influence of environment on mental health, which are collectively referred to as CHARIS-E, with E signifying environment.

The aims of Study 1 and 2 were to: (a) Confirm if individual demographics (age, gender, COVID-19 shielding status (people with specific medical conditions were identified by clinicians as at high risk of severe illness from COVID-19 and were placed on a national ‘shielding list’ and advised to take extra precautions to reduce their risk of infection)) and illness beliefs have a direct association with mental health during the COVID-19 pandemic; (b) determine if the following environmental factors—rurality, area deprivation, access to residential outside space (Study 1), and frequency and duration of visiting green space (Study 2)—have a direct association with mental health during the COVID-19 pandemic; and (c) determine if any of these environmental factors moderate the relationship between individual demographics and illness beliefs and mental health during the COVID-19 pandemic.

2. Materials and Methods

2.1. Design

The CHARIS project was a serial, weekly, nationally representative, cross-sectional, observational study of randomly selected adults in Scotland. CHARIS-E Study 1 draws on data collected for 6 consecutive weeks in June and July 2020, while CHARIS-E study 2 draws on data collected for one week at the beginning of July 2020. These studies were conducted at the point when Scotland had just come out of lockdown [33]. Key contextual changes relevant to this study include the 5-mile restriction on travel distance being lifted (3rd July).

Ethical approval for this study was granted by the Life Sciences and Medicine College Ethics Review Board (CERB) at the University where the Principal Investigator for CHARIS was employed.

2.2. Setting

Scotland has a total population of 5.4 million, 83% of whom are adults. Rural Scotland accounts for 98% of the land mass of Scotland and nearly one-fifth of the population are residents there [34]. Of the population, 70% live in the predominantly urban Central Belt of Scotland. The most deprived areas are in urban areas and the West of Scotland, for example, Glasgow city has 44.4% of the most deprived areas and Aberdeenshire the least, with 2.6% [35]. Whereas c. 65% of Scottish adults live within a 5-minute walk of their nearest green or blue space [36], a smaller proportion of adults in deprived areas live within a 5-minute walk compared to adults in the least deprived areas (58% cf. 68% [37]).

2.3. Participants

All adult men and women aged 16 or older, able to speak English, and currently living in Scotland were eligible to participate. No other exclusion criteria were applied. The CHARIS project was administered by a commercial polling company (Ipsos MORI Scotland) who sampled participants using random digit dialing to landlines and targeted mobiles. Each week, 500 participants were sampled. Quotas were applied to ensure that a representative sample of Scotland adults was achieved. Quotas were based on gender (52% female), age, working status (42% working fulltime), and geographical locations (distribution over the Scottish Parliament regions). A leeway on the quotas (approximately 30%) was allowed to help ensure the overall sample was achieved in a reasonable time.

2.4. Variables and Measures

2.4.1. Psychological Distress (Study 1 and 2)

Psychological distress was measured using the 4-item Patient Health Questionnaire (PHQ-4), which is an ultra-brief screening scale for anxiety and depression [38]. The introductory text was slightly adapted for a telephone as opposed to a written administered survey. Participants were asked: “Over the last 2 weeks, how often have you been bothered by the following problems? Tell me which answer option best applies: (1) Feeling nervous, anxious, or on edge, (2) Not being able to stop or control worrying, (3) Feeling down, depressed, or hopeless, (4) Little interest or pleasure in doing things.” For each item, participants were given the following response options: ‘Not at all,’ ‘several days,’ ‘more than half of the days,’ and ‘nearly every day.’ The total score ranges from 0 to 12, with categories of psychological distress (the term used by PHQ-4 to cover anxiety and depression) being none (0–2), mild (3–5), moderate (6–8), and severe (9–12).

2.4.2. Demographic Variables (Study 1 and 2)

Three sociodemographic variables were measured: (1) Age was assessed continuously in years, (2) gender was assessed using Office for National Statistics binary categories (male, female) [39], (3) shielding category was assessed with the follow item: ‘At the beginning of the pandemic, did you receive a letter or text from your GP or the NHS telling you that you were at risk?’ There were 2 responses: ‘Yes’ or ‘no.’ We did not measure if people did shield, just whether or not they had received a letter or text from government indicating that they should be shielding because they have a medical condition that puts them at risk of severe COVID-19.

2.4.3. Illness Beliefs (Study 1 and 2)

Two illness belief constructs were measured. (1) Threat perception was measured using 2 items to assess the constructs ‘perceived severity’ and ‘perceived vulnerability’: ‘If you were ill with Covid-19 it would be serious for you,’ and ‘It is likely that you will get Covid-19’. There were four responses, ranging from ‘strongly agree’ to ‘strongly disagree’, with the option of responding ‘don’t know’ or ‘prefer not to say.’ In line with the Protection Motivation Theory [40], we multiplied the measures of perceived severity (scale 1–4) and perceived vulnerability (scale 1–4) to produce a total threat perception score (range 1–16). (2) Illness (COVID-19) representations were measured using an adapted brief illness perception questionnaire [41]. The brief illness perception questionnaire uses a single statement to assess constructs from CS-SRM, indicating level of agreement with each statement using a 4-point Likert rating scale. Beliefs about the illness COVID-19 was measured using 5 items: (1) ‘Covid-19 would have major consequences for my life’ (consequences), (2) ‘Covid-19 symptoms would last a long time’ (duration/timeline), (3) ‘You could get Covid-19 again’ (recurrence/timeline), (4) ‘You would spend time worrying about having Covid-19’ (emotional representation), (5) ‘Having Covid-19 would make you feel anxious’ (emotional representation). We calculated the average of these 5 items as a total score for illness representations, with higher scores reflecting more negative representations of COVID-19, Cronbach’s alpha = 0.81 [6].

2.4.4. Environment—Study 1

Three environment variables were measured. (1) Rurality was assessed using an 8-fold urban/rural classification [42]. Participant postcodes were used to classify participants as living in a rural or urban area. The classification is based on size of settlement and drive time to settlements of 10,000 people or more and is as follows: 1 = large urban areas, 2 = other urban areas, 3 = accessible small towns, 4 = remote small towns, 5 = very remote small towns, 6 = accessible rural areas, 7 = remote rural areas, 8 = very remote rural areas. (2) Area deprivation was assessed using the Scottish Index of Multiple Deprivation (SIMD), which looks at the extent to which an area is deprived across seven domains: Income, employment, education, health, access to services, crime, and housing [43]. All 6976 data

zones (postcodes) were grouped into 10 bands (deciles), each containing 10% of the data zones. Decile 1 contained the 10% most deprived data zones in Scotland, and decile 10 contained the 10% least deprived data zones in Scotland. (3) Access to residential outside space was assessed with 1 item: 'Thinking about any outside space you may have at the property you are currently living in (by outside space, I mean a garden, balcony, patio, etc.) do you have access to an outside space at your property?' There were 4 response options which were recoded to assess outside space: The 2 responses, 'yes a private outside space only' or 'yes a private and shared outside space,' were recoded 1, 'yes, a shared outside space only' was recoded 2, and 'No' was re-coded 3. The higher the score, the less access to outside space.

2.4.5. Environment—Study 2

Two variables about green space were measured. (1) Frequency of visiting green space was measured with 1 item: 'On how many days in the last week did you visit any public green or open spaces, for example a park, countryside, wood, play area, canal path, riverside or beach (private or shared garden not included)?', with answer options between 0 and 7 days of the week. (2) Duration of visiting green space was measured with 1 item: 'Thinking about the days in the last week you visited public green or open spaces, how much time, on a typical weekday visit, did you spend in these public green or open spaces?', with 5 answer options: 'Up to 10 min,' '11 up to 30 min,' '30 min up to 1 h,' '1 up to 2 h,' and '2 h or more.' The study also included a potential covariate, namely distance to green space, which was measured with 1 item: 'How far away from your home is your nearest public green or open space?' There were 6 response options: 'Less than a 5 min walk,' 'within a 5–10 min walk,' 'within an 11–20 min walk,' 'within a 21–30 min walk,' 'more than a 30 min walk,' and 'none within walking distance.' These items were based on access to green space indicators in the Scottish Household Survey [37].

2.5. Data Collection

Ipsos MORI administered the self-reported questionnaire by conducting telephone interviews using Computer Aided Telephone Interviewing (CATI).

2.6. Statistical Methods

The data were analyzed using SPSS version 25.0. For all variables, the answers 'I don't know' and 'I prefer not to say' were treated as missing values and, therefore, excluded from the analyses. Most data only had few missing values, which were managed by listwise deletion of cases in any given analyses. p -values of $p < 0.05$ were taken as statistically significant. To test whether the effects of the individual demographic and illness beliefs variables on mental health were moderated by environment variables, moderation models with Hayes' PROCESS macro (v 3.5, model 1) [44] were used. The SPSS macro PROCESS runs each predictor construct individually. The moderation models were tested in 2 steps. In the first step, 2 variables were included, and the effects of demographic independent variables or illness beliefs variables were tested with an environment moderator variable. In the second step, the interaction term between the independent variable and the moderator variable was entered. For the analyses, a 95% bias-corrected percentile bootstrapped confidence interval (CI) method was used, and 5,000 bootstrap resamples were produced for moderation examination. We employed conventional methods for plotting simple slopes to understand moderation effects at 1 standard deviation below and above the mean [45].

3. Results

3.1. Study 1

3.1.1. Participants

Table 1 shows that there were 2969 participants in Study 1, of which 1765 (59.6%) were female, 349 (11.9%) had received a letter or text stating that they should be shielding,

and the median age was 54 years. Overall, this brings all percentages within the quotas, accounting for 30% leeway. Table 1 shows that 1757 (67%) lived in urban areas (1 and 2 classification), 350 (13.3%) lived in small towns (3–5 classification), and 515 (19.6%) lived in rural areas (6–8 classification). In the current study, 133 participants (5.1%) lived in the most deprived areas of Scotland, and 352 (13.4%) lived in the least deprived areas. The overwhelming majority (91.6%) had access to private outside space. Most participants ($n = 2650$; 89.3%) did not report psychological distress.

Table 1. Characteristics of people who participated in Study 1 and 2.

		Study 1		Study 2	
		<i>n</i>	%	<i>n</i>	%
Total <i>n</i>		2969	100	502	100
Demographic variables					
Age (in years)	Median, IQR	54	38–65	53	38–65
Gender	Male	1198	40.4	206	41.1
	Female	1765	59.6	295	58.9
In shielding category	Yes	349	11.9	58	11.6
	No	2592	88.1	442	88.4
Illness beliefs					
Threat perception (range 1–16)	Median, IQR	6.0	4.0–8.0	6.0	4.0–8.0
Illness (COVID-19) representation	Median, IQR	3.2	2.8–3.6	3.2	2.6–3.6
Environmental factors					
Rural Urban classification	1 (large urban area)	901	34.4		
	2	856	32.6		
	3	218	8.3		
	4	84	3.2		
	5	48	1.8		
	6	337	12.9		
	7	93	3.5		
	8 (very remote rural area)	85	3.2		
SIMD	1 (10% most deprived)	133	5.1		
	2	176	6.7		
	3	198	7.6		
	4	216	8.2		
	5	244	9.3		
	6	297	11.3		
	7	317	12.1		
	8	348	13.3		
	9	341	13.0		
	10 (10% least deprived)	352	13.4		
Access to residential outside space					
	Yes, private	2254	91.6		
	Yes, shared only	157	6.4		
	No	51	2.1		
Frequency visiting green space	0 (days per week)			180	35.9
	1			60	12.0
	2			51	10.2
	3			30	6.0
	4			29	5.8
	5			21	4.2

Table 1. Cont.

		Study 1		Study 2	
		<i>n</i>	%	<i>n</i>	%
	6			18	3.6
	7 (days per week)			113	22.5
Distance to green space from home	Less than a 5 min walk			284	56.7
	Within a 5–10 min walk			139	27.7
	Within an 11–20 min walk			52	10.4
	Within a 21–30 min walk			10	2.0
	More than a 30 min walk			7	1.4
	None within walking distance			9	1.8
Duration visiting green space	Up to 10 min			22	6.8
	11 up to 30 min			49	15.2
	30 min up to 1 h			127	39.4
	1 up to 2 h			86	26.7
	2 h or more			38	11.8
Outcome variable					
Psychological distress	Median, IQR	1.0	0.0–3.0	1.0	0.0–3.0
	Normal: Lower than 6	2650	89.3	444	88.8
	Probable case: 6 or higher	316	10.7	56	11.2

Note: Total numbers do not always add up to 2969 and 502 due to missing data. IQR: Interquartile range. SIMD: the Scottish Index of Multiple Deprivation.

3.1.2. Direct Effects of Individual Factors on Psychological Distress

There were significant linear associations between age and psychological distress, with young people more psychologically distressed; gender and psychological distress, with females more psychologically distressed; and shielding category and psychological distress, with people who had received a letter or text stating that they should be shielding being more psychologically distressed (Table 2). There were significant associations between illness (COVID-19) representations and psychological distress and between threat perception and psychological distress, with worse beliefs about the illness COVID-19 and greater threat perception being associated with more psychological distress.

3.1.3. Direct Effects of Environmental Factors on Psychological Distress

There were significant linear associations between rurality and psychological distress, with people living in more rural areas less psychologically distressed; area deprivation and psychological distress, with people from the least deprived areas less psychologically distressed; and access to residential outside space and psychological distress, with people being more psychologically distressed with access to shared outside space or no outside space compared to people with private outside space (Table 2).

Table 2. Linear regression effects of individual demographics and illness beliefs and environmental factors on psychological distress.

			Standardised Beta	R ²	p
Study 1	Individual demographics and illness beliefs	Age	0.16	0.024	<0.001
		Gender [Male dummy coded]	0.14	0.020	<0.001
		In shielding category by the government [Shielded dummy coded]	0.12	0.014	<0.001
		Illness (COVID-19) representation	0.17	0.030	<0.001
		Threat perception	0.13	0.017	<0.001
	Environmental factors	Rurality ^a	0.04	0.002	<0.05
		Area deprivation ^b	0.13	0.016	<0.001
		Access to residential outside space		0.016	<0.001
		Shared outside space	0.11		<0.001
		No outside space	0.07		<0.01
Study 2	Individual demographics and illness beliefs	Age	0.20	0.039	<0.001
		Gender [Male dummy coded]	0.22	0.047	<0.001
		In shielding category by the government [Shielded dummy coded]	0.17	0.028	<0.001
		Illness (COVID-19) representation	0.20	0.041	<0.001
		Threat perception	0.16	0.025	<0.001
	Environmental factors	Frequency visiting green space	0.18	0.031	<0.001
		Duration visiting green space	0.11	0.012	<0.06

^a Eight-fold rural/urban classification. ^b Scottish Index of Multiple Deprivation.

3.1.4. Moderator Effects of Environment on Demographic Variables and Psychological Distress

Rurality did not moderate the relationship between any of the three demographic variables (age, gender, shielding category) and psychological distress (Table 3 and Figure 1). As described above in the Methods section, we used the eight-fold rural/urban classification. We also tested the moderator effect with the six-, three-, and two-fold rural/urban classification [42], which also showed no moderator effects.

Table 3. Moderation of individual demographics and illness beliefs by environmental factors.

		Individual Factors														
		Age			Gender			In Shielding Category by the Government			Illness (COVID-19) Representation			Threat Perception		
		ΔR^2	F, p	simple slope −1sd, 0, +1sd	ΔR^2	F, p	simple slope −1sd, 0, +1sd	ΔR^2	F, p	simple slope −1sd, 0, +1sd	ΔR^2	F, p	simple slope −1sd, 0, +1sd	ΔR^2	F, p	simple slope −1sd, 0, +1sd
Study 1: Environ- mental factors	Rurality ^a		Ns.			Ns.			Ns.			Ns.			Ns.	
	Area deprivation ^b	0.0014	3.93, <0.05	0.02 0.02 0.03	0.0016	4.40, <0.05	0.96 0.74 0.51		Ns.		0.0094	25.21, <0.001	1.03 0.64 0.26		Ns.	
	Access to residential outside space ^c		Ns.			Ns.		0.0075	9.33, <0.001	0.61 2.74 0.32		Ns.		0.0031	3.12, <0.05	0.10 0.23 0.32
Study 2: Environ- mental factors	Frequency visiting green space		Ns.			Ns.		0.0128	6.78, <0.01	0.53 1.52 2.53		Ns.			Ns.	
	Duration visiting green space		Ns.			Ns.		0.0263	9.07, <0.01	3.11 1.52 0.08		Ns.			Ns.	

^a Eight-fold rural/urban classification. ^b Scottish Index of Multiple Deprivation. ^c Reported simple slopes reflect having access to private outside space, shared outside space only, no outside space.

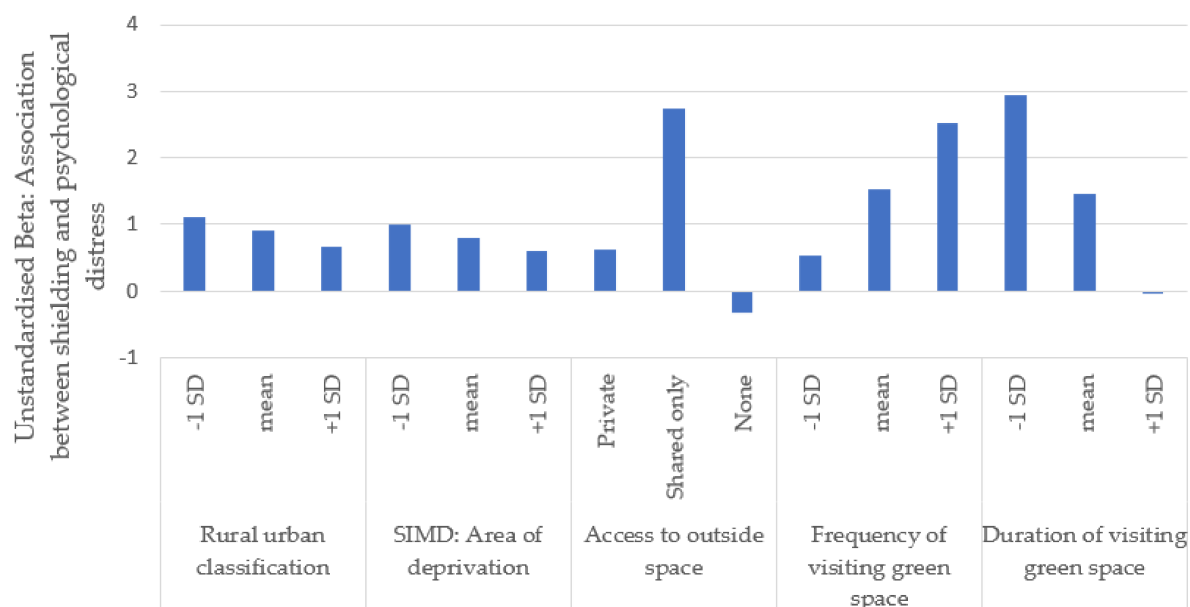


Figure 1. Effects of moderations between shielding category and the environment (Study 1 and Study 2) on psychological distress.

Area deprivation moderated some relationships between demographic independent variables and the dependent variable, psychological distress. Area deprivation moderated the relationship between age and psychological distress. A follow-up simple slope analysis demonstrated that the negative association between age and psychological distress (older people experiencing less psychological distress) was more pronounced in people from the least deprived areas (Table 3). The association between gender and psychological distress was also moderated by area deprivation. The follow-up simple slope analysis demonstrated that males experiencing less psychological distress than females was more pronounced in people from the least deprived areas. Area deprivation did not moderate the effect of being in the group that had received a letter or text stating that they should be shielding and psychological distress.

Access to residential outside space moderated some relationships between demographic independent variables and the dependent variable, psychological distress. The association between age and psychological distress was not moderated by access to outside space, nor was the association between gender and psychological distress. The association between shielding category and psychological distress was moderated by access to shared outside space (Table 3). The follow-up simple slope analysis demonstrated that the effect of receiving a letter or text stating informing the participant that they were in the shielding category on psychological distress as follows: Psychological distress was higher for people who received the letter/text, and this effect was more pronounced for people who had shared outside space compared to people who had access to private outside space. Further, people who had received a letter/text did not have more psychological distress than people who had not received a letter/text among the group who did not have any outside space (see Table 3 and Figure 1).

3.1.5. Moderator Effects of Environment on Illness Beliefs and Psychological Distress

Rurality did not moderate the relationship between illness (COVID-19) representations or threat perception and psychological distress.

Area deprivation moderated the association between illness (COVID-19) representations and psychological distress. The moderation explained a significant increase in variance in psychological distress. A follow-up simple slope analysis demonstrated that

the positive association between illness (COVID-19) representations and psychological distress was more pronounced in more deprived areas than more affluent areas (Table 2). The association between perceived threat and psychological distress was not moderated by area deprivation.

Access to residential outside space did not moderate the association between illness (COVID-19) representations and psychological distress but did moderate the association between threat perception and psychological distress (Table 2). The follow-up simple slope analysis demonstrated that the positive association between threat perception and psychological distress was more pronounced for people with no outside space, followed by people with shared space only and people with private space.

3.2. Study 2

3.2.1. Participants

Table 1 shows that there were 502 participants in Study 2, of which 295 (58.60%) were female, 58 (11.6%) had received a letter or text stating that they were in the shielding category, and the median age was 53 years. Accounting for the 30% leeway, the quotas were met. Just under one-half of participants either did not visit green space or visited only once during a week, and just over one-fifth visited green space every day of a typical week. Over one-half were less than 5 min away from home to green space. The most common duration of time spent in green space was 30 min to 60 min, with 39.4% of participants stating this duration. Most participants ($n = 444$; 88.8%) did not report psychological distress.

3.2.2. Direct Effects of Individual Factors and on Psychological Distress

As per Study 1, there were significant linear associations between age and psychological distress, with young people more psychologically distressed; gender and psychological distress, with females more psychologically distressed; and receiving a letter or text informing the participant that they were in the shielding category and psychological distress, with people who had received a letter/text about shielding being more psychologically distressed. There were also significant associations between illness (COVID-19) representations and psychological distress and between threat perception and psychological distress, with worse beliefs about the illness COVID-19 and greater threat perception being associated with more psychological distress (Table 2).

3.2.3. Direct Effects of Green Space on Psychological Distress

There was a significant linear association between frequency of visiting green space and psychological distress; visiting green space more often was associated with less psychological distress. The association between duration of visiting green space and psychological distress was nonsignificant (Table 2).

3.2.4. Moderator Effects of Green Space on Demographic Variables and Psychological Distress

Frequency of visiting green space did not moderate the effects of age and gender but did moderate the effects of shielding category (Table 2 and Figure 1). The follow-up simple slope analysis demonstrated that the association between receiving a letter or text informing the participant that they were in the shielding category and greater psychological distress was more pronounced in people who more frequently visited green space. In the group of people who visited green space less frequently (1 standard deviation (SD) below the mean), people who had received a letter/text about shielding no longer differed in psychological distress from those who had not received a letter/text about shielding.

Duration of visiting green space did not moderate the effects of age and gender but did moderate the effect of shielding category (Table 2 and Figure 1). The follow-up simple slope analysis demonstrated that the association between receiving a letter/text about shielding and greater psychological distress was more pronounced in people who visited green space for shorter durations. In the group of people who visited green space for longer

durations (1 SD above the mean), people who had received a letter/text about shielding no longer differed in psychological distress from those who had not received a letter/text about shielding.

3.2.5. Moderator Effects of Green Space on Illness Beliefs and Psychological Distress

Neither frequency nor duration of visiting green space moderated the association between both illness (COVID-19) representations and threat perception and psychological distress.

4. Discussion

4.1. Key Results

Whereas the interactions between individual and environmental variables were statistically significant, they explained only a little of the variance in psychological distress. Nonetheless, in each case of statistically significant moderation, being in an area of deprivation, lacking private residential outside space, and being in green space worsened the direct effects of the individual factors. Thus, these environmental factors were not only directly associated with poorer mental health but also appeared to exacerbate the effects of detrimental individual factors. These findings therefore suggest that environment is a significant factor that has influenced mental health during the COVID-19 crisis. However, this interpretation is limited by the cross-sectional nature of the data. For example, it is also possible that people with better mental health made more use of green space.

4.2. Consistency with Other Studies and Implications

The study found that people living in urban areas had worse psychological distress compared to people living in rural areas, thereby supporting previous pre-pandemic research summarised in systematic reviews [19,20] and a study of rural and urban differences in China during the COVID-19 pandemic [46]. However, we found no rural/urban moderation effects. Mental health covers a range of definitions and mental disorders and it may be that psychological distress, which was measured in this study, does not significantly differ between rural and urban areas when other factors are considered, or at least does not in Scotland during the current pandemic at the time when data were collected. Nevertheless, research in Scotland suggests that if there are urban and rural differences in mental health then it is likely to vary by type of mental disorder. For example, one pre-pandemic study found no differences in schizophrenia by degree of urbanisation [47] but other studies found higher rates of death by suicide in rural compared to urban areas [48,49]. Further research focusing on rural and urban differences in mental health during the COVID-19 pandemic should therefore consider focusing on relevant and specific mental disorders, i.e., where rural and urban differences are already known and are likely to be negatively impacted by the COVID-19 crisis. In Scotland, for instance, monitoring rural and urban differences and the impact of the COVID-19 crisis on death by suicide might be relevant. This may be particularly important if the economic impact of the COVID-19 crisis results in unemployment and hardship because there is a concern that the number of deaths by suicide will increase, which could impact rural communities who already have higher incidence of death by suicide, as well as deprived areas that are likely to bear the brunt of an economic recession [50,51].

We found differences in psychological distress in deprived compared to affluent areas and show that females and people with worse illness (COVID-19) representations who live in deprived areas may have worse psychological distress during the COVID-19 crisis. Given that there is strong evidence summarised in systematic reviews that some types of mental health are worse in deprived areas [15–17] and other studies suggesting that mental health during the COVID-19 crisis is worse in deprived areas [5,6] this is not surprising.

Our study suggests that one potential longer-term environmental strategy that could be used to improve mental health during pandemics is improving access to residential outside space. The study, for example, found that mental health was worse for people who

shared outside space who had received a letter/text informing them that they were in the shielding category (incidentally a group at greater risk of poor mental health due to being in quarantine [52]) compared to those who shared an outside space who were not in the shielding category. The results showed that the association between threat perception and mental health was worse for people who did not have access to any residential outside space. UK studies have suggested regional variations in access to residential outside space and use of green space during the COVID-19 crisis [53,54]. In London, 74% of people have access to a garden compared to 97% in Wales, closely followed by 96% of people in the East Midlands [53]. Households with single adults living alone had the least access to gardens (79%) during lockdown compared to households with children or with two or more adults (91%) [53]. These differences in access to residential outdoor space may prove significant for mental health. The importance of the immediate home environment was found in a recent survey of 8177 students from a university institute in an Italian region most heavily hit by COVID-19. The survey found that living in small apartments with poor views was associated with an increased risk of moderate-severe and severe depressive symptoms [55]. The importance of good-quality housing as a contributor to good mental health was found in a systematic review that summarized evidence prior to the pandemic of the relationship between building and health [56]. During the current pandemic, the degree of access people have to outdoor space in the immediate home environment, and whether it is private or shared is therefore important. Given the evidence that living in urban versus rural areas and living in areas of high deprivation are associated with worse psychological distress, those living in homes without outdoor spaces in deprived urban areas may be particularly vulnerable, especially as these locations are also less likely to have public green space within a 5-minute walk [36]. It is difficult in the short term to alter the kind of residential outdoor space that people have access to in existing housing. Therefore, longer-term strategies are needed to effectively address such issues. However, some short-term strategies could be effective during a pandemic, such as offering private access to shared residential space by operating a timed rotation for different households. In some situations, it may also be possible to make adjacent areas of publicly owned outdoor space temporarily available to single households only.

Another potential environment strategy that could be harnessed to improve mental health during the current COVID-19 crisis is improving access to public green space. Research summarized in systematic reviews suggests that adults who live in neighborhoods with a greater quantity of green space around the residential environment report better mental health than adults who live in less green neighborhoods [57]. Our study produces a complex picture in relation to public green space, with greater frequency of visits supporting better mental health in general. However, frequently visiting green space exacerbated the effect of being in the shielding category and worse psychological distress, whereas there were no differences in psychological distress between people who were in and not in the shielding category among those who did not frequently visit green space. One possible explanation is that people in the shielding category who were frequent visitors to green space may experience greater anxiety in public spaces compared to people not in the shielding category because of concerns about being able to keep a 2-meter distance from other people. Of course, if one does not frequently visit green space, then this problem of keeping a 2-meter distance is no longer relevant. Spending only a short duration in green space also exacerbated the effect of being in the shielding category and worse psychological distress, whereas there were no differences in psychological distress between people who were in and not in the shielding category among those who spent a longer in green space. This finding may reflect anxiety associated with being in a public space for people in the shielding category, who might only make short visits because of fear of not being able to adequately distance from other members of the public. Closure of some parks and restrictions on movement might have exacerbated potential problems of crowding and contributed to poorer mental health outcomes during the pandemic [58], particularly for those with no residential outside space. A recent survey found that half

of people in London visited a park or public green space in the past 7 days during April 2020, which was well above the average of 34% across Britain [53]. However, data from a UK-wide survey undertaken under lockdown at the end of April 2020 and in Scotland from 29 May–5 June 2020 [59] showed similar patterns: More people reduced the time spent outdoors or in green space than increased time there while under lockdown [60]. The most marked reduction in time spent in green and open spaces was among older people, who may be likely to perceive themselves most vulnerable to COVID-19. Nonetheless, the majority of participants (63–65%) reported that green and open space benefitted their mental health during the lockdown period, and this was slightly more likely for females and young adults (25 to 44 years old) and those of a higher socioeconomic grade. Our study was undertaken later on, in early July, when Scotland was undergoing a phased lifting of restrictions on movement. Although 84% of our sample had public green space within a 5–10-minute walk, nearly 36% had not visited green space in the previous week. Given the association with reduced psychological distress for frequency of green space visits, maintaining public open space access during the pandemic is clearly important. Moreover, facilitating its safe use by those in the shielding category, e.g., by having well-managed physical distancing measures enforced, could be an important contribution to mental wellbeing.

4.3. Limitations

Current findings should be interpreted in light of key limitations. The study shows that environment exaggerated the effect of some demographics and illness beliefs on mental health, thereby offering some explanation for the widening health inequalities apparent during COVID-19 and possible opportunities for ameliorating these effects during the current and any future pandemics. Nonetheless, we did not include the full breadth of demographic factors, for example, employment status and social-economic status. The environment only explained a small amount of the variance in mental health, and much of the variance in mental health remains unexplained. Second, the current study used only a brief timeframe to measure associations between environment and mental health. Whether environment can attenuate the effects of the COVID-19 pandemic on mental health in the longer term cannot be deduced and merits further investigation. Third, the use of a cross-sectional design precludes the inference of causal effects among the variables. Inferences about relationships between the variables are therefore premised on theoretical suppositions and previous empirical evidence and cannot be inferred from the actual study data. Hence, prospective, longitudinal studies and, where feasible, randomized experiments, are needed to examine the potential causal effects of environment on mental health during pandemics. Related to this point, we did not examine mechanisms to explain environmental differences and mental health. Future research should consider measuring proposed potential mechanisms such as the effect of environment on mood and recovery from stress [61] and environment effects on social support and interpersonal relationships [62,63]. Both of these potential mechanisms may be pertinent in pandemics due to the effects of communicable disease on stress and on restrictions on social interactions to reduce transmission of the infection. Fourth, while objective measures of rurality and area deprivation were used, we measured perceptions of distance to green space rather than objective measurement. There is (Scottish) evidence that there is limited agreement between perceived and objective measures of distance to green space. How people perceived distance to green space during the pandemic seemed particularly relevant to understanding the effects of access to green space on mental health. Nonetheless, future studies should consider the utility of objective measurements of access to green space [64].

5. Conclusions

Environment is important for mental health. This has implications both for public health and for environmental planning, design, and management, including housing design and public open space provision and regulation. Opportunities exist for supporting mental

health by encouraging the use of existing residential outside space and public green space and facilitating safe use, even for those in the shielding category. Future research about mental health during the current COVID-19 crisis should include the environment as a key concept in conceptual models. A multidisciplinary approach should be used to inform public mental health strategies and research to bring psychological and environmental insights about mental health during pandemics.

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Green space accessibility helps buffer declined mental health during the COVID-19 pandemic: evidence from big data in the United Kingdom

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Given accumulating evidence that highlights the negative effect of the COVID-19 pandemic on public mental health, we examine green space accessibility as a potential mitigator. Based on mobility data from 2 million mobile phone users within London between January 2019 and December 2020, we found that, after the COVID-19 outbreak and during lockdowns, residential neighbourhoods within 800 m of the nearest green space had a higher proportion of green-space travellers (0.9–1.4 percentage points) compared with other neighbourhoods. Next, using multiwave data with a matched sample of 4,998 individuals across towns and cities in the United Kingdom, we demonstrate that individuals who lived close to green spaces experienced much less mental distress than those who lived farther away during lockdown periods. We imply that enhancing green space accessibility for residential neighbourhoods can help citizens become more resilient to future pandemics with mobility restrictions.

Since the first known cases in late December 2019, the COVID-19 pandemic has posed unprecedented challenges for everyone worldwide. Owing to the increased risk to public health, many countries have subsequently adopted policies to reduce mobility, such as lockdowns and safe-distancing measures. Although these policies are effective in alleviating the burden on intensive care units and reducing the number of new COVID-19 infections^{1,2}, recent research has identified many consequent negative outcomes. For example, COVID-19 lockdowns resulted in adversities arising from social isolation, inactivity, decreased family and social support, and financial hardship³. Besides the negative economic and social consequences of population mobility restrictions due to COVID-19 (ref. ⁴), research also highlights the heightened issues of public mental health^{5–10}.

Evidence produced from representative cohort studies provided us with a clear comparison of individuals' pre- and in-pandemic psychological states, which showed heightened psychological distress

and a rise in the proportion of people experiencing significant levels of mental illness^{3–8,10,11}. These findings are not completely surprising, considering similar findings in past epidemic events. For example, individuals experienced symptoms of depression and post-traumatic stress disorder during the quarantine order for the severe acute respiratory syndrome (SARS) epidemic^{12–14}. Nevertheless, given the dilemma between the policy needs of mobility restrictions and their negative outcomes, we urgently need to identify potential tools to mitigate such outcomes and maintain the level of public (mental) health even during lockdowns.

Based on a review of 24 studies, scholars have pointed out a few solutions and highlighted that we should keep quarantine orders as short as possible while providing quarantined individuals with more information and support (for example, reducing boredom with improved communication) and encouraging people to participate in voluntary precaution and mobility restriction¹⁵. Although the

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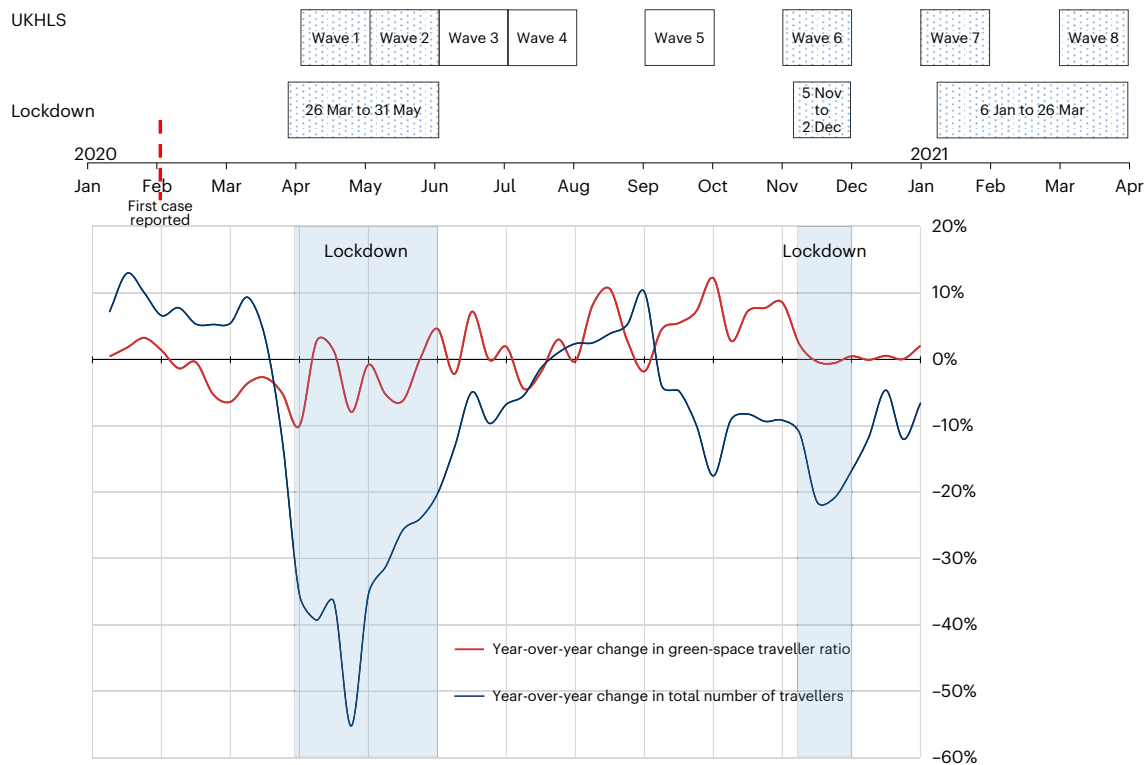


Fig. 1 | Differences in travellers per week in 2020 compared with the corresponding week in 2019 in London alongside the timeline of COVID-19 lockdowns and UKHLS waves. For the UKHLS, each wave of the survey was conducted for 1 month and some waves overlapped with lockdown periods (dotted boxes). With the mobility data available for the period of January 2019

to December 2020, we calculated the change in the number of travellers in each week of 2020 relative to the same week of 2019 (year-over-year change). We did the same calculation for the ratio of travellers to green spaces out of total travellers.

investigation of these quarantine orders is important, lockdown policies (that is, stay-at-home orders) and compulsory social-distancing measures pose a substantial threat to a much larger number of people as their daily activities are fully or partially restricted. Therefore, it is still important for us to understand how we can successfully mitigate the detrimental effect of these measures.

To accomplish this goal, we explore research in environmental psychology, particularly on the effect of green space exposure on public mental health. Exposure to green spaces have been widely recognized to promote mental health and wellbeing for a variety of populations in different circumstances^{16–19}. Previous research suggests negative associations between green space exposure and psychological distress^{20–23}. Accumulating research has also pointed out the potential beneficial effects of green spaces during COVID-19, when the mobility of the general public was restricted on a compulsory or voluntary basis^{24–26}. Research has also shown that people changed the amount of time spent visiting green space and urban green infrastructure after COVID-19 and during lockdowns^{27–29}. To contribute to this booming literature, this study seeks to better understand how the change in public mobility is related to the beneficial effects of green spaces on mental health during the COVID-19 lockdowns. We aim to adopt a more precise way of documenting the ecological-level mobility information of residents during COVID-19 lockdowns. Doing so can enable us to accurately observe mobile activities in a much smaller area compared with other methods (for example, Google Mobility Reports)²⁸.

Research has shown that living closer to green spaces supports mental and general health and also helps prevent depression in young adults¹⁷. COVID-19 lockdowns highlighted the need for outdoor walks to nearby parks. In fact, Google searches for ‘go for a walk’ significantly increased right after lockdown orders in many countries³⁰. However, the surge in interest in going for short walks is not always matched by

the supply of green spaces (for example, parks and gardens), especially in high-density urban areas^{31,32}. This highlights another important question: Is the mental health of some individuals more adversely affected by the COVID-19 pandemic and lockdowns than others due to unequal access to green spaces? A recent *BMJ* commentary highlighted the importance of equal access to green spaces during the COVID-19 pandemic and called for more academic and political attention to this issue³³. Research over the past three years has responded and started to pay attention to the relationship between green space accessibility and mental health since the outbreak of COVID-19. From the Web of Science database, we found 241 empirical articles from various subject fields, including public health, environmental studies, urban studies and planetary studies. Some have used panel data to examine the relationship, surveying individuals’ green space activities and mental status^{34,35}, and others have tried to examine general mobility patterns during the COVID-19 pandemic³⁶.

There is, however, little empirical testing that combines rigorous mobility data with panel data of a relatively large, representative sample using a validated mental health scale. By adopting mobility data from 2 million mobile phone users and the multiwave longitudinal survey data, this study presents a thorough empirical examination of the relationship between mobility to green spaces and mental health status after COVID-19 and during lockdowns. Furthermore, our research highlights accessibility to green spaces as a crucial feature for both green-space travel and mental wellbeing, which has important implications for not only mental health services but also urban planning and policymaking. We choose to examine this question in London, United Kingdom (UK), as it provides us with a great opportunity to carry out large-scale, quasi-experimental analyses by uniquely utilizing mobile-phone-based mobility data and multiwave surveys during and after lockdown periods.

Table 1 | Descriptive statistics of travel data in London

Variables	LSOAs within 800 m of the nearest green space (treated LSOAs)		LSOAs farther from the nearest green space (comparison LSOAs)	
	Mean	s.d.	Mean	s.d.
Total number of travellers during non-work hours	1,316.92	1,559.24	1,234.88	1,431.24
Total number of green-space travellers during non-work hours	78.33	287.04	36.27	102.09
Proportion of green-space travellers during non-work hours	5.96%	10.95%	2.90%	2.90%
Proportion of green-space travellers before COVID-19	5.67%	5.67%	3.01%	4.84%
Proportion of green-space travellers after COVID-19	6.31%	12.85%	2.77%	5.70%
Proportion of green-space travellers during non-lockdown periods after COVID-19	6.26%	12.07%	2.84%	2.84%
Proportion of green-space travellers during lockdown periods after COVID-19	6.46%	14.94%	2.58%	6.46%
Demographic characteristics (2019, 2020)				
Population	1,867.71	441.81	1,849.58	527.33
Male	45.62%	49.81%	43.15%	49.53%
Under 18 years old	21.50%	5.24%	23.69%	4.78%
18–39 years old	35.22%	10.19%	32.35%	8.77%
40–59 years old	26.28%	3.82%	26.46%	3.12%
60–89 years old	16.33%	5.99%	16.82%	6.10%
Over 89 years old	0.68%	0.60%	0.67%	0.58%
White	73.44%	15.31%	68.50%	17.94%
Asian	15.80%	13.57%	20.72%	16.94%
Black	6.44%	5.36%	7.43%	5.99%
Other races	4.30%	2.39%	3.33%	2.11%
Socioeconomic characteristics (English Indices of Deprivation 2019)				
Index of Multiple Deprivation score	20.67	10.77	22.16	10.97
Income score (rate)	0.13	0.08	0.14	0.07
Employment score (rate)	0.08	0.05	0.09	0.05
Education, skills and training score	10.25	8.84	15.31	10.26
Health deprivation and disability score	−0.44	0.79	−0.34	0.63
Crime score	0.27	0.58	0.25	0.56
Barriers to housing and services score	30.03	9.18	32.89	9.99
Living environment score	31.81	10.96	27.15	10.37
Income Deprivation Affecting Children Index score	0.17	0.10	0.18	0.09
Income Deprivation Affecting Older People Index score (rate)	0.23	0.14	0.23	0.13
Number of LSOAs	2,149		2,686	
Number of LSOAs × weeks (96 weeks)	206,063		257,198	

Non-work hours include 16:00–24:00 on working days and 08:00–24:00 on weekends and bank holidays. The English Indices of Deprivation 2019 comprises seven different domains that are combined and weighed to generate the LSOA's Index of Multiple Deprivation score out of 100. A higher score means more deprivation. Income Deprivation Affecting Children Index and Income Deprivation Affecting Older People Index are supplementary indices for income. For more detailed descriptions of the data, refer to online guidance⁶¹.

Results

We used a sample of mobile location records from 4,835 residential lower layer super output areas (LSOAs, defined as neighbourhoods in this paper), which are the smallest geographic units available for major UK administrative data and contain about 1,500 residents on average. Figure 1 shows that the total number of travellers within London significantly decreased after the COVID-19 outbreak by 25% compared with the pre-COVID-19 period. The most significant drop of 55% was seen when the first lockdown was imposed. Interestingly, even during lockdowns, more people tended to travel to green spaces than to other spaces while general mobility showed a significant downward trend. Starting from April 2020, the probability that an individual would travel to green space relative to other places increased compared with the same week in 2019. We attempt to perform spatial analyses and identify

which LSOAs within London experienced higher or lower increases in the proportion of green-space travellers during lockdowns compared with the average amount in 2019. Extended Data Fig. 1 shows that LSOAs in red (those with increasing probability of travel to green spaces during lockdowns compared with pre-COVID-19 periods) mainly cluster around major green spaces in London. This implies that LSOAs closer to green spaces were much more likely to see an increase in the proportion of travellers to green spaces than other LSOAs.

Table 1 provides summary statistics for the subsamples of LSOAs whose edges were within 800 m of a green space edge (treated LSOAs) and those further away from green spaces (comparison LSOAs). The probability that people residing in treated LSOAs travel to green spaces relative to non-green spaces within London was double compared with residents in other LSOAs. This difference in travel behaviours by

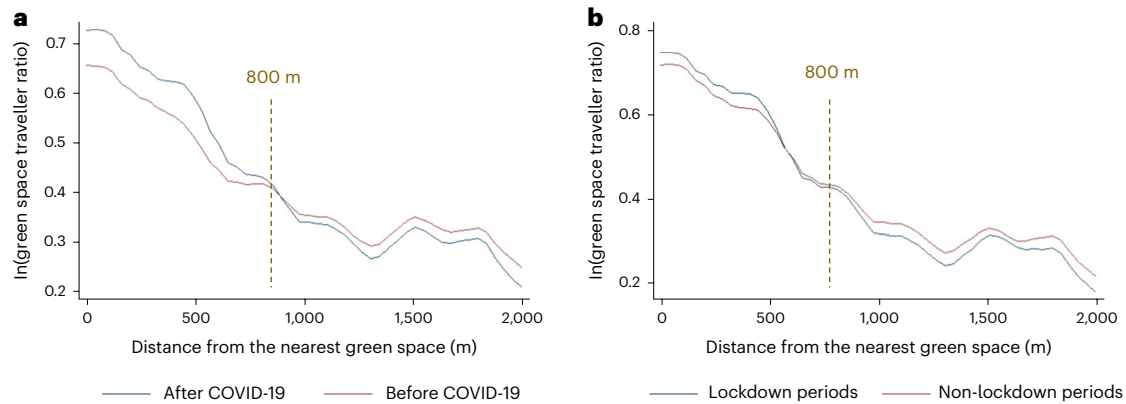


Fig. 2 | Green-space distance gradient for the ratio of green-space travellers during non-work hours in London. a, Green-space traveller ratio before and after COVID-19 outbreak. **b,** Green-space traveller ratio during non-lockdown and lockdown periods

proximity to green spaces was most significant when people were stuck at home. After the COVID-19 outbreak, more residents in the treated LSOAs travelled to green spaces while fewer in the comparison LSOAs did so. During the lockdowns, more people living closer to green spaces travelled to green spaces while fewer living farther did so, compared with non-lockdown periods. Figure 2 shows consistent results, with the green space distance gradient for the proportion of green-space travellers. While demographic characteristics were quite similar for the treated and comparison LSOAs, residents in treated neighbourhoods were more likely to be white. While the treated LSOAs were less deprived in general than comparison neighbourhoods, they were slightly worse in terms of health desirability, crime and living environment, potentially because planned green spaces are more likely to be located in higher density areas.

Our empirical model results report the effect of being located closer to green spaces on the probability of travelling to green spaces. Cross-sectional estimates show pre-existing conditions that the proportion of green-space travellers within London is about 1.4 percentage points higher in neighbourhoods within 800 m of the nearest green space than in neighbourhoods located farther (Table 2). If we only account for post-COVID-19 periods, the difference in this proportion is slightly larger at 1.6 percentage points, implying the increasing importance of the distance to green spaces for the travel patterns to green spaces. Table 2 also shows the estimation results of the difference-in-difference (DID) specification that adds the temporal variations (before versus after the COVID-19 outbreak and lockdown versus non-lockdown). In general, the proportion of travellers to green spaces relative to other places decreased by 0.3 and 0.2 percentage points after the COVID-19 outbreak in London and during lockdowns. However, the probability of travelling to green spaces from LSOAs that were closer to green spaces was approximately 0.9 percentage points higher compared with other London neighbourhoods with similar characteristics but located farther from green spaces. During the lockdowns, these neighbourhoods experienced an additional increase of 0.5 percentage points in the proportion of green-space travellers. Given that the average proportion of green-space travellers in London was 4.2%, the total increase of 1.4 percentage points during lockdown periods is significant. Additionally, the increase is much greater than the general reduction in the proportion of green-space travellers after COVID-19 and during the lockdowns (0.3 and 0.2 percentage points, respectively). Our results suggest that green space accessibility plays an even more important role in travel to green spaces when people face strict restrictions on movement. Note that for our robustness test, we use the ratio of green-space travel out of total travel counts and the results are consistent (Extended Data Table 1).

After establishing the evidence on the significant role of proximity to green spaces in the higher probability of travelling to green spaces

after the COVID-19 outbreak and during lockdowns, we next analysed the mental distress of residents. From a sample of 4,998 individuals matched from the original sample of 19,020 survey respondents from the UK Household Longitudinal Study (UKHLS), Table 3 shows that people reported higher mental distress during the lockdowns compared with post-COVID-19 non-lockdown periods. In particular, respondents living farther than 800 m from green spaces (comparison group) experienced a more substantial increase in their mental distress score during the lockdowns. Their mental distress score was 0.5 higher on average during the lockdowns compared with post-COVID-19 non-lockdown periods. Those living closer to green spaces (treatment group) experienced an average increase of 0.35 in the score. As individuals in the treatment and comparison groups showed quite homogeneous characteristics after matching (Table 3), this difference in the change in mental distress scores is likely to be attributable to the proximity of one's residence to green spaces. The treated individuals tended to earn more than their matched counterparts even after matching and this is potentially because green spaces in our data are planned spaces that are more likely to be concentrated in cities with higher productivity.

Table 4 shows that the mental distress score is not statistically different for individuals residing within 800 m of the nearest green space (treatment group) and their matched counterparts who have similar attributes but reside farther from green spaces (comparison group). These results with only the spatial variation suggest a null effect of the accessibility to green spaces on mental health in pre-existing conditions. When we use the DID specification, adding the temporal variation between lockdowns and non-lockdowns, the difference in the average mental health scores between the treatment and comparison groups remains statistically insignificant at the 10% level. Then, results report that the mental distress score is 0.605 higher during lockdowns compared with non-lockdown periods, which is substantial given the average score is only 2.31 for our matched sample. Nonetheless, individuals residing close to green spaces were able to offset this increased mental distress score by 0.106 more than those in the comparison groups. Finally, we performed a robustness test with the matched subsample of London residents. The results demonstrate that the increase in the mental distress score during lockdowns is even higher for London residents (0.734 for London versus 0.605 for the UK). Also, green space accessibility plays a more important role in reducing this score for London residents (-0.378 for London versus -0.106 for the UK). These results suggest that while the urban environment with higher density may be more vulnerable to mental distress during lockdowns, enhancing green space accessibility could mitigate such a risk. In addition, we observe that treated individuals had less volatile psychological distress scores between lockdown and non-lockdown periods than their matched counterparts. In particular, in the last wave of the UKHLS, which fell in a lockdown period, the

Table 2 | Empirical estimation of the impact of proximity to green spaces on the proportion of green-space travellers

	Cross-sectional			DID			Cross-sectional (post-COVID-19)			DID (post-COVID-19)		
	Coef.	s.e.m.	P	Coef.	s.e.m.	P	Coef.	s.e.m.	P	Coef.	s.e.m.	P
LSOAs located within 800 m of the nearest green space	0.014	0.002	0.000	0.010	0.002	0.000	0.016	0.003	0.000	0.015	0.002	0.000
After COVID-19 outbreak				−0.003	0.000	0.000						
Lockdown										−0.002	0.000	0.000
LSOAs within 800 m of the nearest green space and after COVID-19				0.009	0.001	0.000						
LSOAs within 800 m of the nearest green space and lockdown										0.005	0.001	0.000
Observations	463,261			463,261			212,411			212,411		
R ²	0.227			0.232			0.237			0.237		
Demographic characteristics	Yes			Yes			Yes			Yes		
Socioeconomic characteristics	Yes			Yes			Yes			Yes		
Local authority fixed effects	Yes			Yes			Yes			Yes		
Month fixed effects	Yes			Yes			Yes			Yes		

Standard errors (s.e.m.) are clustered at the LSOA level. The analysis uses two-sided statistical tests.

treatment group showed a much lower median mental distress score than the comparison group, suggesting that people living closer to green spaces were able to better stabilize their mental status during lockdowns (Extended Data Fig. 2).

Discussion

This study first examined how London's population mobility to green spaces changed after COVID-19 outbreak and during lockdowns and then explored how better access to green spaces could affect psychological distress with a longitudinal cohort sample during lockdowns. The overall number of travellers within London has significantly decreased after the COVID-19 outbreak as compared with the same weeks in 2019, and the most significant drop was seen when the first lockdown was imposed. This trend was similarly discovered by many studies in other contexts. For example, the average time spent in non-residential locations decreased by 40% in response to various mobility restriction policies across 80 countries globally³⁷. Our study built an empirical model that considers population mobility patterns derived from anonymous mobile phone data. This approach is arguably better than using other available mobility data (including air and rail travel records, GPS loggers, Google records, apps or other social media sources) as the latter could only capture the trajectories of subpopulations that use specific transport tools or mobile applications³⁸. By contrast, the comprehensive coverage of mobile phone users aged 15–65 years with a market share of 25% can help obtain a more representative sample for the whole population in London, and an accurate reflection of movement patterns between their residences and high-frequency destinations. Additionally, this study integrated location information of registered green spaces in London and examined whether the antenna polygon of the travel destination of mobile phone users from different residential neighbourhoods overlaps with green spaces. Such investigation unveiled that an individual's probability to travel to green spaces rather than other places has increased compared with the same period in 2019, and the tendency to increase travel to green spaces continued during lockdowns even when general mobility decreased.

While the COVID-19 outbreak has substantially affected population mobility patterns within London owing to people's voluntary precautionary behaviours, lockdown orders have brought an even higher reduction in mobility. When people were ordered not to leave home without a reasonable excuse, they immediately reduced more

than half of their travel activities^{39,40}. Even with this reduced mobility, the probability of travelling to green spaces relative to other places showed a quick recovery about one week after the first lockdown order. Additionally, lockdown measures appeared to have different effects on populational mobility across different stages of the outbreak. In the period of our sample, we observed a much higher reduction in the year-over-year mobility changes during the first lockdown order than during the second lockdown. A potential reason could be that the second order (27 days) was much shorter than the first (91 days), and moving around would have been a more appealing option if allowed during the first order when the weather was more suitable for movement and outdoor activities (that is, summertime) than the second (that is, wintertime). In addition, people likely adjusted their travel behaviours from the continuing pandemic by the time they reached the second order. In terms of travel to green spaces, we found a more stable trend during the second lockdown, which could again be attributable to travel behaviour adjustment. The probability of travelling to green spaces instead of other places during the second lockdown is consistently higher than in the same weeks in 2019.

More importantly, this study examined how proximity between residence and green spaces can affect individuals' mobility during lockdowns. After the COVID-19 outbreak and during the lockdowns, individuals who lived close to green spaces were more likely to visit those spaces than other non-green spaces. These findings echo the discoveries from some of the urban sustainability and environmental studies. For example, previous studies found that stressed individuals like to access green spaces more than other spaces⁴¹, that neighbourhood greenery can help to facilitate social support⁴², and that publicly accessible neighbourhood nature can be associated with residents' increased sense of community belonging, which in turn improves mental health outcomes⁴³. Additionally, a previous review in environmental research has also pointed out the benefits of accessing green space on improving wellbeing, by reducing exposure to environmental stressors, restoring capacities, and building capacities¹⁹. Recently, urban research also showed that green infrastructure across cities can interplay with respondents' residential locations, as well as their socio-demographic profiles and lockdown policies, to predict residents' outdoor recreation behaviour⁴⁴. In fact, the unequal access to green spaces presents a troubling picture to policymakers, as individuals who live more than 800 m away from green spaces tended to travel less to green spaces

Table 3 | Descriptive statistics of matched survey data in the UK

Variables	Individuals living within 800 m of the nearest green space (treatment)		Individuals living farther from the nearest green space (comparison)	
	Mean	s.d.	Mean	s.d.
Total score of mental distress	2.36	3.32	2.26	3.29
Total score of mental distress during lockdown periods	2.49	3.33	2.45	3.35
Total score of mental distress during non-lockdown periods	2.14	3.30	1.95	3.17
Demographic characteristics				
Age	54.21	16.94	54.96	16.33
Gender (male versus female)	42.80%	49.48%	42.07%	49.37%
White	86.88%	33.76%	90.92%	28.73%
Black	2.50%	15.63%	1.82%	13.38%
Number of household members	1.63	1.25	1.64	1.22
Number of children	0.38	0.80	0.35	0.73
Having a partner	71.43%	45.17%	71.50%	45.14%
Economic characteristics				
Household monthly earning	£4,189.67	£14,503.48	£3,415.81	£12,780.11
Individual monthly earning	£1,716.01	£6,298.52	£1,414.13	£5,310.86
Having financial difficulties	3.81%	19.15%	4.58%	20.90%
Health status				
Having long-term health issues	39.95%	48.98%	41.35%	49.25%
COVID-19 infection	0.67%	8.16%	0.76%	8.67%
Number of days doing moderate activities	2.69	2.60	2.61	2.57
Moderate/heavy drinker	46.42%	49.87%	46.54%	49.88%
Smoker	6.70%	25.01%	7.23%	25.90%
Number of individuals	2,496		2,496	
Number of individuals × survey waves	15,529		15,730	

Treatment and comparison groups for this matched sample are derived by the PSM procedures. The purpose of PSM is to ensure individuals in treatment and comparison groups to be highly homogeneous with respect to their demographic, economic, and health attributes.

than those who have better access. This by itself might not be an issue as the mobility restriction measures were meant to reduce social interactions and population mobility. However, this study highlighted the potential issue in regard to impaired psychological wellbeing.

By using the longitudinal household sample in the UK to track temporal changes in national mental health from before COVID-19 to the subsequent lockdown period, this study examined how individuals' psychological wellbeing was affected during the pandemic with detailed time series data. Unlike previous inquiries with similar data^{6,45}, we associated the survey data with the individual's residence information and further investigated whether living close to green spaces helped individuals battle the negative influence of lockdown on their psychological wellbeing. Similar to a previous study⁶, we found that individuals were significantly distressed during the lockdowns (versus non-lockdown periods) after accounting for all relevant factors.

In particular, we observed a 0.605 increase in the psychological distress score.

Supporting our main hypothesis, we found that, during lockdowns, individuals who lived close to green spaces (that is, within 800 m) saw a much smaller increase in the distress score than those who lived farther away after controlling for all other potential determinants of mental health known in the literature. We also found that mental distress states, as represented by the General Health Questionnaire scores, were much more stable for individuals who lived close to green spaces than those who lived farther away. This is particularly interesting as we identified a potential group of the population that had a higher volatility of psychological distress during lockdowns. Unlike prior studies that focused on examining effects of the individual characteristics—such as gender; age; educational attainment and socioeconomic position; Black, Asian, and minority ethnic backgrounds; and living conditions (for example, living alone)^{46–49}—this study identified an environmental factor that can provide policymakers with an opportunity to intervene. Building from the seminal work that has shown the positive effect of simply having a window view of a natural setting on the speed of recovery and quality of postoperative experiences⁴⁹, this study suggested that residing in a place close to public green spaces could also have a significantly positive impact on individuals' mental health, especially when their mobility is restricted by lockdown orders.

Additionally, lockdown measures during the COVID-19 pandemic provided a better context to examine the effects of green space accessibility than quarantines in previous epidemics. For the majority of residents in London during lockdowns, certain travel outside of the home was permitted so they could travel to green spaces. Because quarantines of previous epidemics posed stronger mobility restrictions and the number of affected individuals was typically smaller than an entire city lockdown, they provided no opportunity to observe public mobility, especially to green spaces⁴⁵. To our knowledge, this is one of the first studies to combine public mobility data with a longitudinal household survey and systematically examine how lockdowns affect the level of public mental distress with a focus on exposure to green spaces. The unique datasets enabled the long-term tracking of public mobility and mental health before and during COVID-19. Although we didn't test this proposition directly for those who cannot travel outside of their residences at all during lockdowns (for example, someone with quarantine orders), we have reason to believe that providing a place with a view of green spaces⁵⁰, incorporating vertical greenery (that is, the integration of vegetation onto the vertical structures of buildings)⁵¹, or even a plasma display of nature⁵², could be beneficial for the mental wellbeing of such individuals.

Overall, our findings echo recent research on how local green spaces can help to promote citizen mobilities during COVID-19 (ref. ⁵³), and demonstrate a higher level of aggravating psychological distress for those who cannot access green space easily (that is, those living in areas not within walking distance). As countries and cities around the world face the risks of future lockdowns, these findings emphasize the importance of supporting and paying attention to individuals who have inferior access to green spaces. Some cities like Paris and Singapore have already begun plans to enhance accessibility to green spaces for more residential neighbourhoods and they are likely to be in a good position to prevent excessive mental illnesses during similar future pandemics^{54,55}. Besides creating more quality green spaces, policymakers can consider expanding trails and green networks to provide better environments for walking to larger green spaces and to ensure more equal access to green spaces for all citizens during lockdowns. It is key to supporting more vulnerable groups during the current pandemic and those in the future.

This work has limitations, which could spark future research. First, because of the data limitations, our two analyses were performed with different samples and at different geographic scales (that is, mobility analysis in London and mental distress analysis in the UK). Ideally,

Table 4 | Empirical estimation of impact of living closer to green spaces on psychological distress

	Cross-sectional			DID (all UK respondents)			DID (London respondents)		
	Coef.	s.e.m.	P	Coef.	s.e.m.	P	Coef.	s.e.m.	P
Living within 800m of the nearest green space	0.071	0.072	0.324	0.129	0.080	0.109	0.405	0.251	0.106
Lockdown				0.605	0.059	0.000	0.734	0.190	0.000
Living within 800m of the nearest green space and lockdown				−0.106	0.050	0.035	−0.378	0.163	0.021
Observations	31,259			31,259			3,314		
R ²	0.097			0.112			0.134		
Demographic characteristics	Yes			Yes			Yes		
Economic characteristics	Yes			Yes			Yes		
Health status	Yes			Yes			Yes		
City fixed effect	No			Yes			Yes		
Wave fixed effect	No			Yes			Yes		

Standard errors (s.e.m.) are clustered at the LSOA level. The analysis uses two-sided statistical tests. The specification includes the wave fixed effects to make adjustment for multiple companions across different survey waves.

we would have wanted to obtain the travel information of individual survey respondents so that we could directly associate their travel patterns to green spaces with their mental wellbeing. Next, although we believe that we have done our best to identify individual mobility and associate that with green space location, there could still be some miscalculation on the location of the individuals given that the size of the antenna could potentially cover areas with both green and non-green spaces. Finally, our analyses on mental health do not fully account for potential confounders related to the built environment such as walkability, bike-ability, and the level of noise, which could be important for the mental health of residents, especially when they stay longer in their residences after COVID-19 and during the lockdown. Future research would benefit from further investigating clearer underlying mechanisms through which green spaces mitigate the negative impacts of COVID-19 on mental health. Also, as lockdown measures vary significantly by country and by region, more comparative analyses would be useful to generalize empirical results.

Methods

Ethical regulations

The study was conducted in compliance with the EU General Data Protection Regulation and the internal privacy policies of Telefónica. Ethics approval for the COVID-19 web and telephone surveys (ETH1920-1271) in UKHLS, a major data source for this study, was granted by the University of Essex Ethics Committee, which also approved out use of this data.

Data sources

Our data come from several sources (Extended Data Fig. 3). The first is a dataset on the weekly bilateral flow information of 2 million mobile phone users from their residence (origin) to destinations within London over the period January 2019 to December 2020. We were granted access to anonymized mobility data from Telefónica that contain aggregated counts of travel between antenna points and do not include any personally identifiable information. The data consist of travel counts and travel frequency between antenna points within London as well as the time spent in the destination; stays were longer than 1 hour to exclude temporary movements by car and/or public transport. Existing research using mobile phone data suggests 1–1.5 hours as the threshold for stable detection of location changes^{56,57}. Additionally, as the difference in the travel counts between 30 min and 1 h was not significant in our data, we believe that the 1-hour threshold is a reasonable choice and safer from possible noise. In our analysis, we define ‘travel’ as an event where a mobile phone user connects to a non-home antenna for over 1 hour, using pre-identified information about the home antenna.

This means that we do not account for the antennas connected on the way to the destination antennas. Our data include travel information for both work hours (08:00–16:00 on working days) and non-work hours (16:00–24:00 on working days and 08:00–24:00 on weekends and bank holidays). To perform the analysis with resident characteristics at the temporally stable geographic level, we perform spatial interpolation from the origin–destination flows between antenna points to flows between the residential LSOAs and green/non-green space destinations. A more detailed explanation of the mobility data processing is presented in Supplementary Discussion 1.

Second, we marked the location information of a broad range of open spaces with good vegetation coverage including registered public parks, cemeteries and town squares in London provided by Historic England in August 2021 (ref. ⁵⁸). The data contain 1,699 entries in England and 168 listings in London as of August 2021. The advantage of using these data is that we could control for the level of attractiveness, accessibility and maintenance of green spaces that are open to the public by focusing on registered green spaces instead of small, unregistered spaces that may be exclusive or specialized (for example, community gardens and golf courses). More detailed explanations for our choice of green spaces are provided in Supplementary Discussion 2. By identifying whether the destination locations belong to these green spaces based on the interpolated mobility flows, we obtained the ratio of the number of green-space travellers relative to the number of total travellers from each residential LSOA in a given week. We use this ratio during non-work hours for our main analyses because travel to green spaces is a recreational activity that does not occur frequently during work hours. We demonstrate that results using the ratio of green-space travel out of total travel counts are consistent with those using this main measure. To account for demographic and socioeconomic characteristics at the LSOA level, we use the LSOA-level data on population characteristics for 2019 and 2020 provided by the Office for National Statistics⁵⁹, as well as 2020 data on ethnic composition from the Consumer Data Research Centre, which was obtained with special permission⁶⁰. We also use the 2019 English Indices of Deprivation for the 4,835 LSOAs in London provided by the Ministry of Housing, Communities and Local Government⁶¹. The data include diverse neighbourhood indicators on income, employment, education, health, crime, barriers to housing affordability and local services, living environment, and income deprivation affecting children and older people.

Our final dataset was generated from the UKHLS in 2020 and 2021, provided by the University of Essex, to assess individuals’ psychological distress levels^{62,63}. The data contain various information on respondents such as age, gender, race, family composition, income

and physical health conditions across eight different waves of COVID-19. After cleaning up the data with no or missing responses, we have unique responses of 19,020 residents across cities and towns in the UK. The basic nature of this survey is a longitudinal study that follows the same sample of people over time, although each wave adds a small number of people into the sample with cross-sectional sampling weights. For our main analyses, we use the panel sample with longitudinal sampling weights. We also received special permission to obtain the residence LSOA information of respondents, which enables us to measure the distance of one's residence to the nearest green spaces to approximate their probability to travel to green spaces. As only 2,073 survey respondents reside in London, and this number becomes even smaller after matching, we use the UK sample for our main analyses and perform the robustness test with the London subsample. Full details on the recruitment, sampling, retention, and weighting of the sample are available in the UKHLS User Guide⁶².

Identification strategies

Figure 1 shows how lockdown periods overlap with our datasets of mobile-phone-based mobility in London as well as the UKHLS. During our sample period, the UK government imposed three national lockdowns that placed restrictions on movement; no person was allowed to leave the place where they live without a reasonable excuse. Lockdown laws in the UK encompass restrictions on movement and gatherings, as well as the closure of and restrictions on businesses. As restrictions on movement are the strongest regulation that affects travel behaviours, we focus on these phases for our research and call them 'lockdowns'⁶⁴. During these lockdowns, non-essential street businesses (for example, cafes, restaurants, bars and pubs) were closed and people were asked to stay at home as much as possible and were strictly banned from gathering. People were, however, still allowed to do outdoor exercises either alone or with other household members or seek medical assistance.

For our mobility analysis at the neighbourhood level, we examine the threshold distance to green spaces that determines the significant changes in travel patterns after the COVID-19 pandemic (versus before) and during the lockdown (versus non-lockdown). We utilize straight-line distances between the edges of green spaces and residence LSOAs using a Geographic Information System. Figure 2 presents the gradient of the proportion of the number of green-space travellers out of all travellers by the distance to the nearest green space. The distinct divergence at around the 800 m radius suggests that people residing within 800 m of green spaces show different changes in travel patterns to green spaces compared to those who live farther away. The 800 m cut-off coincides with a walkable distance in the existing literature⁶⁵. As this is identified as an ideal spatial treatment, we generate a binary variable that captures whether the nearest green space is located within 800 m of each residential LSOA.

Next, to minimize potential confounding issues and provide a causal interpretation for our model of the role of proximity to green space to mental distress, we adopt a standard logic of a counterfactual causal inference design. Consistent with the above analysis on neighbourhood-level mobility, we identify green space proximity using a distance limit of 800 m for each survey respondent. Our potential treatment group comprises all UKHLS respondents residing in LSOAs within 800 m of the nearest green space, and those who reside in LSOAs that are farther belong to the potential comparison group. Among the pool of survey respondents in the potential comparison group, we select the closest match for each individual residing within 800 m of the nearest green space by using a propensity score matching (PSM) procedure. Our final sample size after matching is 2,496 individuals for the treatment group and 2,496 for the comparison group, both of which are highly homogeneous with respect to age, race, family composition, earnings and drinking/smoking habits (see Supplementary Discussion 3 for the detailed matching process and the quality of the matched sample).

Finally, we pay attention to UKHLS data where psychological distress was measured using 12 questions from the General Health Questionnaire, a well-validated tool used to screen and diagnose generalized anxiety disorder in clinical practice and research⁴⁷. It uses a four-point scale, where higher point values indicate a more deteriorated condition ('not at all'/'same as usual' were given a score of 0; 'more than usual'/'much more than usual', a score of 1). We then convert the total score into the measure with 12 points (asymptomatic [score 0], sub-clinically symptomatic [1–3], symptomatic [4–6] and highly symptomatic [7–12])⁶⁶.

Empirical models

We adopted a DID approach using variations in the distance from green spaces and in pre/post periods of the COVID-19 outbreak or lockdown event. We employed the DID model as follows:

$$V_{jt} = \beta D_{jt}^{800} + \theta D_{jt}^{800} \times \text{Post}_{jt} + \delta \text{Post}_{jt} + X'_j \gamma + \varphi_m + \alpha_l + \varepsilon_{jt}. \quad (1)$$

where V_{jt} is the proportion of travellers to green spaces out of all travellers in LSAO j in week t , D_{jt}^{800} is a binary indicator of whether there is a green space within an 800 m radius of LSAO j , and Post_{jt} is a binary indicator that travel behaviours in week t occurred after the COVID-19 outbreak or during lockdown periods. θ picks up how the proportion of green-space travellers in LSOAs located within 800 m of the nearest green space changes after COVID-19 or during lockdown periods compared with the pre-pandemic or non-lockdown periods. X is a control vector of LSAO-specific characteristics such as population size, gender (men versus women), racial group (White, Asian, Black, others), age group (under 18 years, 18–29, 30–45, 46–59, 60–89, over 89), as well as a wide range of neighbourhood indices related to income, employment, education, health disability, crime, barriers to housing and services, and living environment. We include monthly fixed effects and local authority fixed effects to control for unobserved heterogeneity. ε_{jt} is an error term. To account for the potential serial correlation of residuals within an LSAO, we cluster standard errors at the LSAO level. Results of this model are presented in Table 2.

Next, we attempted to compare the level of mental distress between individuals residing in LSOAs that are within 800 m of the nearest green spaces and those residing in farther LSOAs but within the same city. We employed a DID model similar to equation (1) as follows:

$$M_{iw} = \beta D_{iw}^{800} + \theta D_{iw}^{800} \times \text{LD}_{iw} + \delta \text{LD}_{iw} + X'_i \gamma + \varphi_w + \alpha_c + \varepsilon_{iw}. \quad (2)$$

where M_{iw} is the mental distress score of individual respondent i from the matched sample in wave w , D_{iw}^{800} is a binary indicator of whether there is a green space within an 800 m radius of the residence of respondent i , and LD_{iw} is a binary indicator that a mental health survey in wave w is conducted during lockdown periods. θ picks up how the mental health score changes for those who reside within 800 m of the nearest green space during lockdown periods. X is a control vector of demographic and economic variables such as age, gender (men versus women), racial group (White, Black, other), number of household members, number of children, whether living with a partner, household/individual monthly earning, and whether having financial difficulties as well as health status such as whether having long-term health issues, whether infected with COVID-19, number of days doing moderate activities, whether drinking moderately or heavily, and whether smoking. We include wave fixed effects and city fixed effects to control for unobserved heterogeneity and uneven distribution of green spaces across cities. ε_{iw} is an error term. Results of this model are presented in Table 4.

Statistics and reproducibility

No statistical method was used to predetermine the sample size for the mobility data; the sample size was based on the availability of

network access logs from Telefónica. Similarly, the original sample size for the individual survey data was based on UKHLS that represents the UK population (for more details of the UKHLS sampling strategy, refer to User Guide⁶²). As mentioned above, we use the PSM procedures to derive treatment and comparison groups from this original sample so that these groups share similar attributes. Data preparation was carried out using Python, R package (version 4.1.0) and ArcGIS Online. Regressions were performed with the STATA statistical package (version 14.0). Heat map representation was performed in ArcGIS Online.

Consent

We do not use any personally identifiable data, so informed consent is not needed.

Reporting summary

Further information on research design is available in the Nature Portfolio Reporting Summary linked to this article.

Data availability

The mobility dataset is proprietary to Telefónica and the dataset is subject to strict privacy regulations. The mobility dataset was anonymized and aggregated before being shared with the authors. The dataset could be available on request after a non-disclosure agreement is signed and discussed. Contact details for accessing the data are on

/. The UKHLS is a study by Understanding Society, an initiative funded by the Economic and Social Research Council and various UK government departments. To access the UKHLS data⁶³ with residential LSOA information of survey respondents, one needs to receive permission from the data owner, the University of Essex, through the special licence application process with the UK Data Service. The 2020 data on ethnic composition was obtained with special permission from the Consumer Data Research Centre. Other datasets, including the location of green spaces and LSOA-level neighbourhood attributes, are publicly accessible^{38,39,61}.

Code availability

The custom code that supports the findings of this study is available from the corresponding author upon request.

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Author contributions

K.O.L., K.M.M. and S.P. co-designed the study. K.O.L. and S.P. collected the necessary datasets while S.P. was responsible for processing the mobility and UKHLS data. K.O.L. undertook data analyses and interpretation of the results. K.O.L. and K.M.M. wrote the report, and all authors critically reviewed the manuscript.

Competing interests

The authors declare no competing interests.

Additional information

Extended data is available for this paper at [REDACTED]

Supplementary information The online version contains supplementary material available at [\[REDACTED\]](#)

Correspondence and requests for materials should be addressed to Kwan Ok Lee.

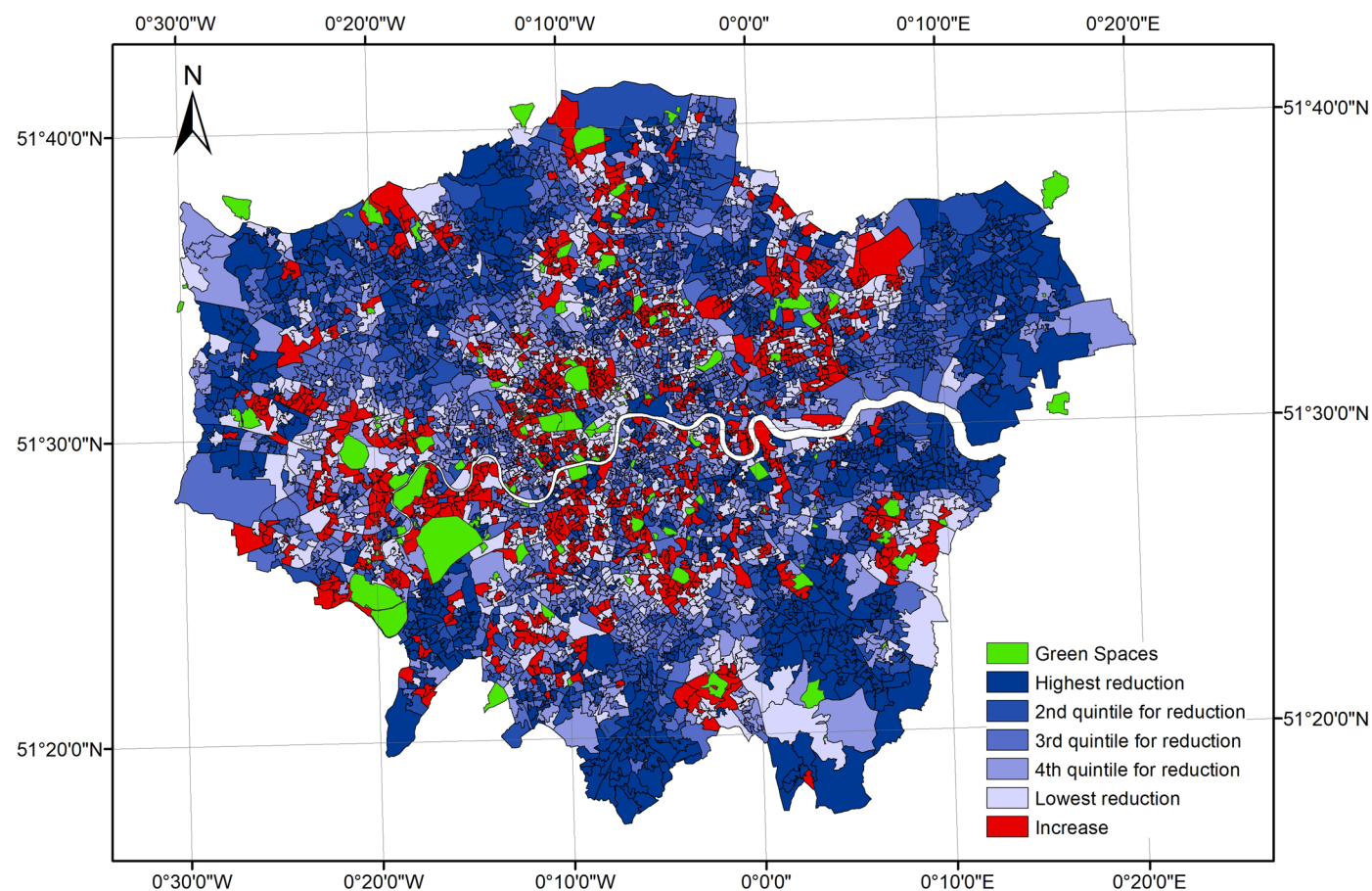
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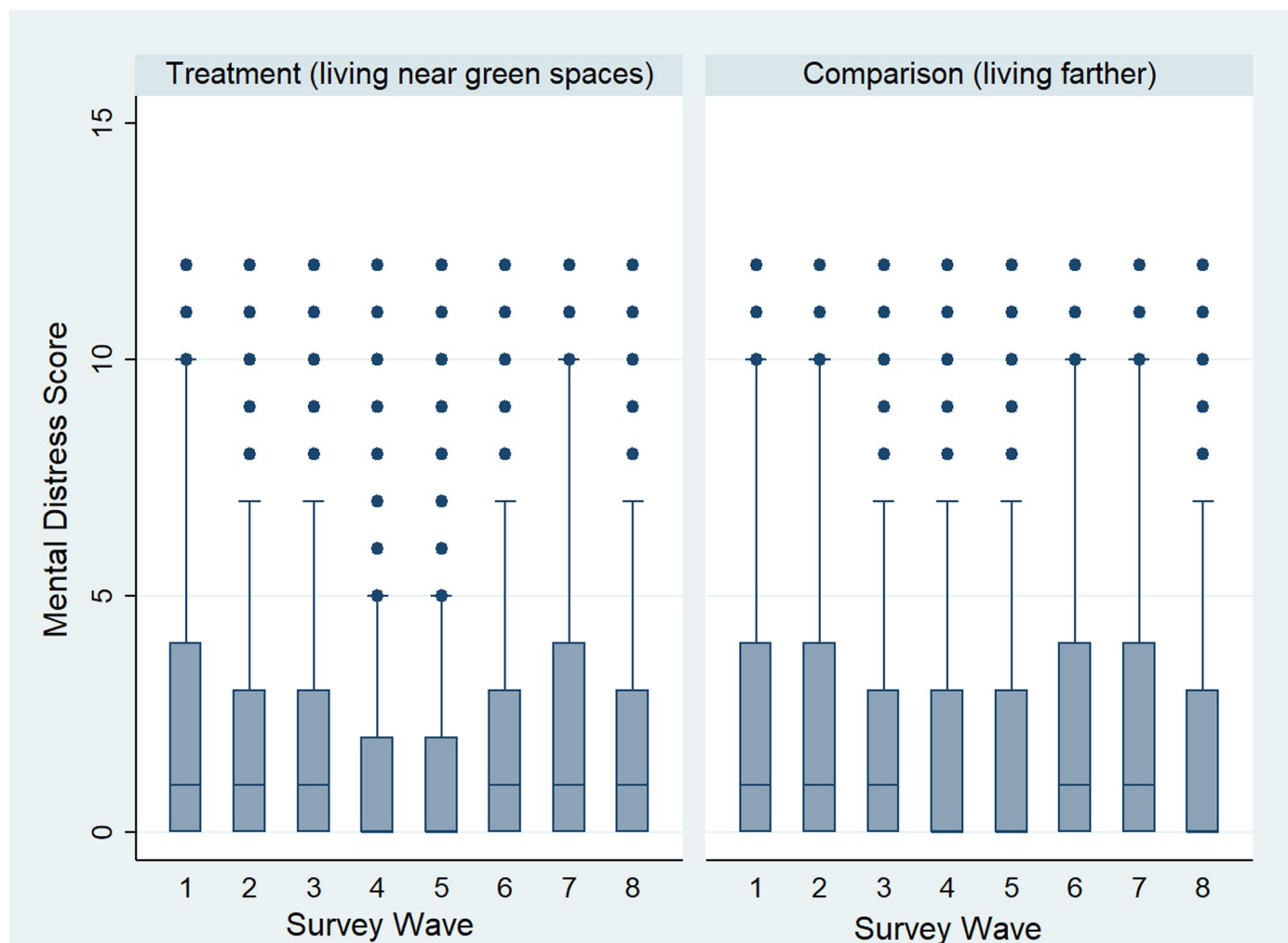
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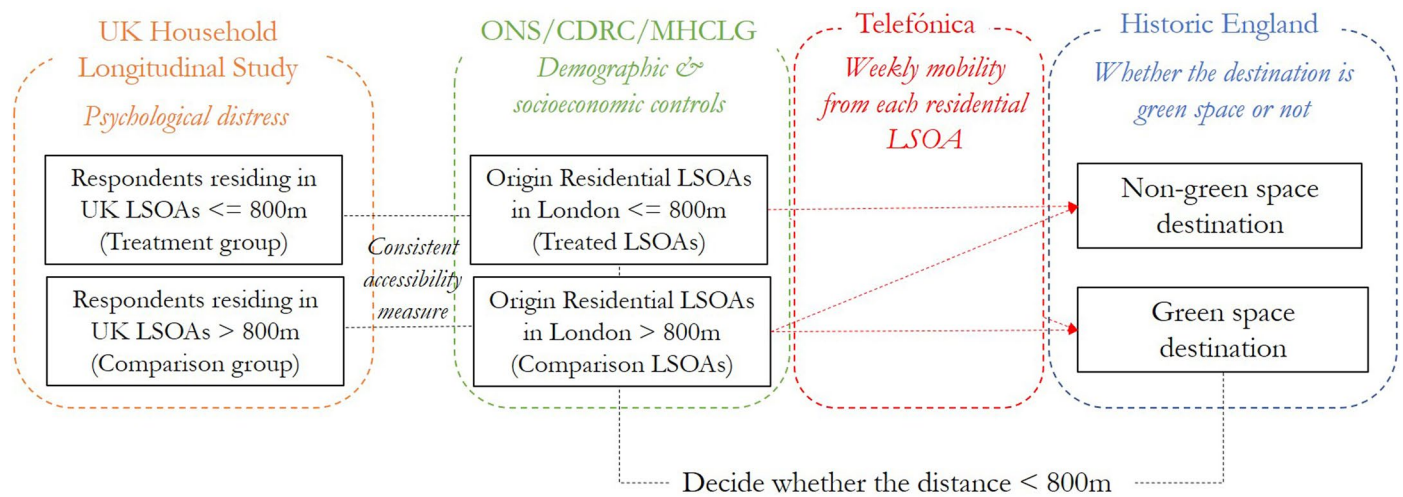
Extended Data Fig. 1 | Changes in share of number of travellers to green spaces during lockdowns compared to share of green-space travellers during pre-COVID-19 periods. The blue colours represent LSOAs that experienced the reduction in the share of the number of travellers to green spaces out of total travellers during the lockdowns when compared to pre-COVID-19 periods. Darker

blue areas experienced a higher reduction. On the other hand, the red colour represents LSOAs that experienced the increase in the share of green-space travellers during the lockdowns when compared to pre-COVID-19 periods. Each of the six categories with different colours have approximately the same number of LSOAs.



Extended Data Fig. 2 | Boxplot of mental distress score by different waves for treatment versus comparison groups. Note that Waves 1, 2, 6, 7, and 8 fall into the lockdown periods in the UK. Here, we did not derive results from technical replicates because each wave belongs to the different time periods (lockdown versus non-lockdown) and this temporal variation is a key for our analysis. N (Treatment) = 2,496 individuals observed over 8 independent waves;

N (Comparison) = 2,496 individuals observed over 8 independent waves. Boxplot minimum is the smallest value within 1.5 times interquartile range below 25th percentile, maximum is the largest value within 1.5 times interquartile range above 75th percentile. Centre is the 50th percentile (median), box bounds 25th and 75th percentile.



Extended Data Fig. 3 | Summary of data and sources. This summarizes the data sources, key indicators, and their links. ONS: Office for National Statistics; CDRC: Consumer Data Research Centre; MHCLG: Ministry of Housing, Communities and Local Government.

Extended Data Table 1 | Robustness test for Table 2 with the travel frequency measure (rather than number of travellers). Standard errors (s.e.m.) are clustered at the LSOA level. The analysis uses two-sided statistical tests.

Dependent Variable	(1)			(2)			(3)			(4)		
	Proportion of Green Space Travels			Proportion of Green Space Travels			Proportion of Green Space Travels (post-COVID-19 periods)			Proportion of Green Space Travels (post-COVID-19 periods)		
	Coef.	S.E.	P	Coef.	S.E.	P	Coef.	S.E.	P	Coef.	S.E.	P
LSOAs located within 800 m of the nearest green space	0.023	0.004	0.000	0.020	0.003	0.000	0.023	0.004	0.000	0.023	0.004	0.000
After COVID-19 outbreak				-0.003	0.001	0.000						
Lockdown										-0.001	0.000	0.013
LSOAs within 800 m of the nearest green space & After COVID-19				0.005	0.001	0.000						
LSOAs within 800 m of the nearest green space & lockdown										0.002	0.001	0.011
Observations	463,261			463,261			212,411			212,411		
R-squared	0.143			0.143			0.150			0.150		
Demographic characteristics	Yes			Yes			Yes			Yes		
Socioeconomic characteristics	Yes			Yes			Yes			Yes		
Local authority fixed effects	Yes			Yes			Yes			Yes		
Month fixed effects	Yes			Yes			Yes			Yes		

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|-------------------------------------|--|
| ☐ | ☒ The exact sample size (n) for each experimental group/condition, given as a discrete number and unit of measurement |
| ☐ | ☒ A statement on whether measurements were taken from distinct samples or whether the same sample was measured repeatedly |
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| ☐ | ☒ A full description of the statistical parameters including central tendency (e.g. means) or other basic estimates (e.g. regression coefficient) AND variation (e.g. standard deviation) or associated estimates of uncertainty (e.g. confidence intervals) |
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First, the mobility dataset is proprietary to Telefónica and the dataset is subject to strict privacy regulations. The mobility dataset was anonymised and aggregated before being shared with the authors. The dataset could be available on request after a Non-Disclosure Agreement is signed and discussed. Contact details for accessing the data are on <https://www.telefonica.com/en/sustainability-innovation/innovation/telefonica-research/>. Next, to access the UKHLS data63 with residential LSOA information of survey respondents, one needs to receive permission from the data owner, the University of Essex, through the special license

application process with the UK Data Service. Also, the 2020 data on ethnic composition was obtained with special permission from the Consumer Data Research Centre. Other datasets including the location of green spaces and LSOA-level neighbourhood attributes are publicly accessible.

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All studies must disclose on these points even when the disclosure is negative.

Study description	Our study examines the association between green space accessibility and mental health outcomes during the COVID-19 pandemic.
Research sample	We used several existing datasets to conduct our research. First, we used the UK Household Longitudinal Study that represents the population of the UK (https://beta.ukdataservice.ac.uk/datacatalogue/studies/study?id=8644). We received permission to use this data with residential LSOA information of survey respondents through the special license (Project No. 212669). Next, the mobility dataset is proprietary to Telefónica and the dataset is subject to strict privacy regulations. The mobility dataset was anonymised and aggregated before being shared with the authors. The dataset could be available on request after an NDA is signed and discussed. Finally, other data that we used are all aggregate data. We use the LSOA-level data on population characteristics for the years of 2019 and 2020 provided by the Office for National Statistics (source: https://www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/populationestimates/datasets/lowersuperoutputareamidyearpopulationestimates), as well as the 2020 data on ethnic composition from the Consumer Data Research Centre (https://data.cdrc.ac.uk/dataset/cdrc-modelled-ethnicity-proportions-la-geography), which was obtained with special permission. We also use the 2019 English indices of deprivation for the 4,835 LSOAs in London provided by the Ministry of Housing, Communities and Local Government (source: https://www.gov.uk/government/statistics/english-indices-of-deprivation-2019).
Sampling strategy	For the UK Household Longitudinal Study, we copy the sample description from User Guide: "These samples are probability samples of postal addresses. In England, Wales and Scotland they are clustered and stratified, in Northern Ireland unclustered systematic random samples. Northern Ireland and areas in Great Britain with high immigrant and ethnic minority populations were oversampled." For more details please refer to User Guide available at https://doc.ukdataservice.ac.uk/doc/8644/mrdoc/pdf/8644_ukhls_covid19_user_guide_v10.0.pdf . For the mobility data, the sample size was based on the availability of network access logs from Telefónica.
Data collection	For the UK Household Longitudinal Study, according to User Guide, the survey was conducted either by phone or the web. The survey data was collected irrelevantly from our study hypotheses. This is same for the mobility dataset provided by Telefónica.
Timing	The survey was conducted from Apr. 2020 to Sep. 2021.
Data exclusions	No data from the original survey sample were excluded.
Non-participation	No participant dropped out.
Randomization	Randomization is not relevant to our study as we did not perform any experiment.

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Data Article

OakEcol: A database of Oak-associated biodiversity within the UK



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ABSTRACT

Globally there is increasing concern about the decline in the health of oak *Quercus* trees. The impact of a decline in oak trees on associated biodiversity, species that utilize oak trees, is unknown. Here we collate a database of all known birds, bryophytes, fungi, invertebrates, lichens and mammals that use oak (*Quercus petraea* and *Q. robur*) in the UK. In total 2300 species are listed in the database. For each species we provide a level of association with oak, ranging from obligate (only found on oak) to cosmopolitan (found on a wide range of other tree species). Data on the ecology of each oak associated species was collated: part of tree used, use made of tree (feeding, roosting, breeding), age of tree, woodland type, tree form (coppice, pollarded, or natural growth form) and season when the tree was used. Data on use or otherwise by each of the 2300 species of 30 other tree species was also collated. A complete list of data sources is provided. For further insights into how this data can be used see Collapsing foundations: The ecology of the British oak, implications of its decline and mitigation

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options [1]. Data can be found at EIDC <https://doi.org/10.5285/22b3d41e-7c35-4c51-9e55-0f47bb845202>.
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Specifications table

Subject area	Ecology
More specific subject area	Species level ecology
Type of data	Database (Microsoft Office Access Database) but held in EIDC as CSV files with details as to how to reconstruct as a relational database.
How data was acquired	Collation of data across literature and databases
Data format	Collated in standardized format from many data sources
Experimental factors	The species included in the database were known to occur in the UK and with evidence that the species uses oak. The quality of the data used was categorised as peer-reviewed (which included quality-controlled databases) or non-peer-reviewed and if the data came from a UK source or a non-UK source.
Experimental features	The data was collected under a common, predefined template, to ensure the same data was collected across taxon groups. A literature review, that including searching relevant databases of species records, was conducted to identify which species used oak trees and if they used any of 30 other tree species.
Data source location	UK
Data accessibility	Mitchell, R.J.; Bellamy, P.E.; Ellis, C.J.; Hewison, R.L.; Hodgetts, N.G.; Iason, G.R.; Littlewood, N.A.; Newey, S.; Stockan, J.A.; Taylor, A.F.S. (2019). Oak-associated biodiversity in the UK (OakEcol). NERC Environmental Information Data Centre. https://doi.org/10.5285/22b3d41e-7c35-4c51-9e55-0f47bb845202
Related research article	Mitchell RJ, Bellamy PE, Ellis CJ, Hewison RL, Hodgetts NG, Iason GR, Littlewood NA, Newey S, Stockan JA Taylor AFS. (2019) Collapsing foundations: the ecology of the British oak, implications of its decline and mitigation options. Biological Conservation https://doi.org/10.1016/j.biocon.2019.03.040

Value of the data

- The most complete list of biodiversity associated with oaks (*Quercus*) in the UK.
- The data can be used to identify which species are most at risk of declining, if oak declines.
- The information could be used to prioritize conservation of species at a range of scales.
- The database can be used to identify other tree species that oak-associated species would use instead of oak. Such information may be used in management plans of woods and forests.

1. Data

There are 24 tables in a relational database (Fig. 1, Table 1) containing information about species that use oak (*Quercus petraea* and *Q. robur*) in the UK. The primary table (Species) lists all the oak associated species and information on their conservation status. A secondary table “Only oak associated species” is a subset of this table excluding the species that use the oak woodland habitat but not specifically the oak tree. There are tables detailing the age of tree used, the tree form used (coppiced, pollard, natural), the part of the tree used, and the season when the tree is used. Information on how the species uses the tree (for feeding directly or indirectly or as a habitat for living or nesting/roosting) is collated in the “Use” table. Information was also collected on the type of oak woodland the species uses, specific relationships between invertebrates and fungi and other ecological notes. The “Alternative Tree” table provides information on if the oak associated species are known to use any of 30 other alternative tree species, with a related table giving the tree species names in both English and Latin. There are also 10 definition tables providing definitions for the terms used in the other tables. The final table is a list of references used in collating the database, these references can be linked to the data in the other tables.

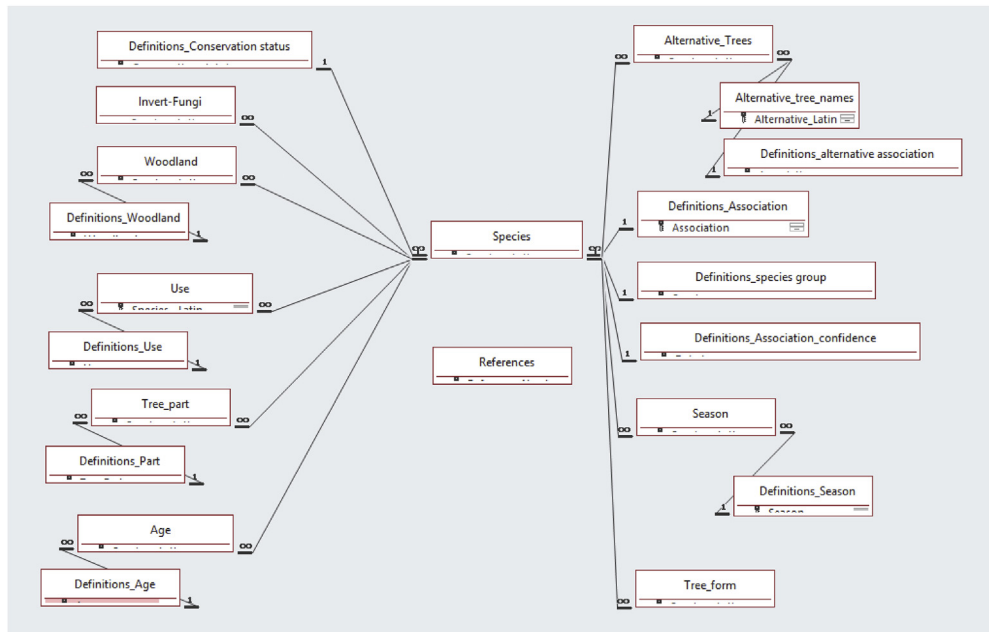


Fig. 1. Diagram of the relationships between the different tables within the database OakEcol.

All the data sources are classified as peer-reviewed (or quality-controlled database) or non-peer-reviewed and either from the UK or not.

2. Experimental design, materials, and methods

A literature review was conducted in 2016/17 using an extensive search of both published and grey literature, together with unpublished information, in order to identify as comprehensively as possible those species that use *Q. petraea/robur* (oak-associated species) and the nature of this association in the UK. We confined ourselves to six taxon groups: birds, bryophytes, invertebrates, fungi, lichens and mammals, and did not attempt to include algae, bacteria or other micro-organisms. All data were collected under a pre-defined common structure and collated into a relational database (OakEcol).

The level of association of each species with *Q. petraea/robur* within the UK was recorded as 'obligate', 'high', 'partial', 'cosmopolitan' or 'uses' (Table 2). For some guilds it was possible to define these categories based on the number of records of a species occurring on *Q. petraea/robur* as opposed to other tree species. For other guilds a more subjective assessment was made based on the available literature (Table 3).

Information on the conservation status of associated species was recorded (Table 4); this was broadly grouped as to whether the species was known to have some form of conservation protection, as different taxon groups have different measures of conservation protection. Where relevant, according to taxon group, additional information was collated on the IUCN status of the species, if the species was on a UK Red Data book list, a Biodiversity Action Plan species or on the priority species list of any of the four countries within the UK. Birds species were also recorded as red, amber or green on the UK Birds of Conservation Concern list. The conservation status of the species and its level of association with oak were combined to provide an "at risk to oak decline" category for each species ranging from RED-coded which was high risk to GREEN-coded, no risk [1].

Table 1
List of Tables in the OakEcol database and a description of the data they contain.

Table name	Description of data
Species	The main data table in the database to which most other tables are related. A list of all oak-associated species, Latin and English names, their level of association with oak, their conservation status, if the species is native to the UK. Information on if the species is a priority species in any of the four countries within the UK, a red data book species, a BAP species or an IUCN species. A reference number, related to the reference table is included. This table includes some mammal species that are found in oak woods but don't directly use the oak tree.
Only oak tree species	As per the species table but only includes those species that use oak trees, not just the woodland habitat. Also those fungi that are only recorded once, and then on oak are removed as there is no reliable way of knowing their level of association with oak.
Age	The ages of the oak tree that the oak associated species uses
Alternative_tree_names	A list of Latin and English tree names for species that could replace oak and for which an assessment as to whether the oak associated species do or not use them, was made.
Alternative_Trees	Information on if the oak associated species will or will not use 30 other tree species
Definitions_Age	Definitions of the age classes used in the Age table
Definitions_alternative association	Definitions of the level of association classes used when assessing how the oak associated species uses the 30 alternative tree species
Definitions_Association	Definitions of the classes of association that the oak associated species have with oak.
Definitions_Association_confidence	Definitions of the classes of data describing the data used to assess the level of association of the oak associated species with oak and the alternative tree species. These classes were based on if the data was peer-reviewed or not and if it came from the UK or not
Definitions_Conservation status	Definitions of the conservation status of a species used in the Species table.
Definitions_Part	Definitions of the tree parts used in the Tree_parts table
Definitions_Season	Definitions of the seasons used in the Seasons table
Definitions_species group	Definitions of the species groups used in the species table
Definitions_Use	Definitions of how the oak associated species use the oak tree, as used in the Use table.
Definitions_Woodland	Definitions of woodland types as used in the Woodlands table.
Ecology_notes	Notes on the ecology of the oak associated species.
Invert-fungi	Information on if the oak associated invertebrate eats fungi or carries a fungus and aids its spread.
Only_Woodland	A list of those species (mammals) which are associated with oak woodlands but do not directly use the oak trees.
References	Full details of all references used.
Season	The season when the oak associated species uses the oak tree.
Tree_form	The tree form (pollard, coppice and/or natural growth form) that the oak associated species uses.
Tree_part	The part of the tree that the oak associated species uses.
Use	How the oak associated species uses the oak tree e.g. feeding, roosting, nesting.
Woodland	The type of woodland in which the oak associated species uses oak trees.

Table 2
Criteria used to assess the use of *Quercus robur/petraea* in the UK. This includes species dependent on other species that use *Q. robur/petraea*, such as parasites and some of the predatory insects.

Association with oak in UK	Definition
Obligate	Unknown from other tree species. NOTE: when dealing with parasites the following criteria were used: obligate host + obligate parasite = obligate; obligate host + parasite with multiple hosts = uses; highly associated host + obligate parasite = highly associated; highly associated host + parasite with multiple hosts = uses
High	Rarely uses other tree species
Partial	Uses oak more frequently than its availability
Cosmopolitan	Uses oak as frequently or lower than availability
Uses	Uses oak but the importance of oak for this species is unknown

Table 3

Methods used to assess level of species associated with *Q. robur/petraea*. For each species the reference used is listed in the OakEcol database.

Species group	Data sources and criteria used to assess association
Lichens	For all lichen species which had been confirmed as recorded on <i>Q. robur/petraea</i> within the British Lichen Society database (1960–2017), the number of times that each species had been recorded on <i>Q. robur/petraea</i> as a proportion of the total number of all records across all substrata (including corticolous, terricolous and saxicolous records, etc) was calculated. The ‘level of association’ for a species was considered <i>obligate</i> if 100% of records were from <i>Q. robur/petraea</i> , <i>high</i> if >75% of records were from <i>Q. robur/petraea</i> , <i>partial</i> if >50% of records are from <i>Q. robur/petraea</i> , and <i>cosmopolitan</i> if the number of records from <i>F. excelsior</i> trees <50%.
Bryophytes	The British Bryological Society (BBS) records and the following references [2–8], were the primary sources used. Additional literature was used for specific species (see OakEcol database)
Fungi	The species assessed was limited to the fungal taxa reported in The Fungal Records Database of Britain and Ireland (FRDBI) http://www.fieldmycology.net/FRDBI/FRDBI.asp which contains a total of 51914 post-1960 records where the associated organism was <i>Quercus</i> or Oak. Many of these were erroneously included within the field of fungi – with many being lichens. Detailed assessments of fungi were limited to those with 98–100% of records with <i>Quercus</i> and which were considered to be obligate associates and those with 80–97% of records with <i>Quercus</i> and which were considered to be highly associated with <i>Quercus</i> . Entries with a species epithet suggesting a strong affinity with <i>Quercus</i> were also assessed.
Invertebrates	The principal data sources used were [9–20]. Additional literature searches were carried out for specific species (see OakEcol for details).
Mammals	The handbook of British Mammals [21] retrieved from http://books.google.co.uk/books?id=w_UJNAAACAAJ was used as the main information supplemented with additional literature searches for specific species (see OakEcol for details).
Birds	The assessment of birds associated with <i>Q. robur/petraea</i> trees was primarily based on online searches of peer reviewed literature. Further information was sought from unpublished reviews on the habitat associations and requirements for woodland birds. See OakEcol for complete list of references.

Table 4

Definitions of criteria used to record the conservation status of the oak-associated species within the OakEcol database.

Measure	Category	Definition
Conservation concern	Conservation concern-protected.	Species is listed as being of some conservation concern in one of BAP, Red data, IUCN, Birds of conservation concern
	Conservation concern-unprotected.	Known by experts to be of conservation concern but it is currently not protected i.e. it is not listed in one of BAP, Red data book, IUCN, birds of conservation concern
	Not of Conservation Concern.	There is enough data to know that the species is of no conservation concern
	Unknown	The distribution/population of the species is unknown and it is not known whether this species should be of conservation concern
IUCN	Endangered	A taxon is Endangered when the best available evidence indicates that it meets any of the criteria A to E for Endangered, and it is therefore considered to be facing a very high risk of extinction in the wild. http://www.iucnredlist.org/static/categories_criteria_3_1
	Vulnerable	A taxon is Vulnerable when the best available evidence indicates that it meets any of the criteria A to E for Vulnerable (see Section V), and it is therefore considered to be facing a high risk of extinction in the wild. http://www.iucnredlist.org/static/categories_criteria_3_1
	Near Threatened	A taxon is Near Threatened when it has been evaluated against the criteria but does not qualify for Critically Endangered, Endangered or Vulnerable now, but is close to qualifying for or is likely to qualify for a threatened category in the near future. http://www.iucnredlist.org/static/categories_criteria_3_1
	Least concern	A taxon is Least Concern when it has been evaluated against the criteria and does not qualify for Critically Endangered, Endangered, Vulnerable or Near Threatened. Widespread and abundant taxa are included in this category. http://www.iucnredlist.org/static/categories_criteria_3_1

(continued on next page)

Table 4 (continued)

Measure	Category	Definition
	Data deficient	A taxon is Data Deficient when there is inadequate information to make a direct, or indirect, assessment of its risk of extinction based on its distribution and/or population status. http://www.iucnredlist.org/static/categories_criteria_3_1
BoCC4	Red	Birds of conservation concern - red listed
	Amber	Birds of conservation concern - amber listed
	Green	Birds of conservation concern - green listed
Priority sp_England	Yes	Listed as a Priority species in England (species of principal importance) http://webarchive.nationalarchives.gov.uk/20140711133551/http://www.naturalengland.org.uk/ourwork/conservation/biodiversity/protectandmanage/habsandspeciesimportance.aspx
	No	Not listed as a Priority species in England
Priority sp_Wales	Yes	Listed as a Priority species in Wales (Section 7 List) http://www.biodiversitywales.org.uk/Environment-Wales-Bill
	No	Not listed as a Priority species in Wales
Priority sp_NI	Yes	Listed as a Priority species in Northern Ireland https://www.daera-ni.gov.uk/articles/northern-ireland-priority-species
	No	Not listed as a Priority species in Northern Ireland
Priority sp_Scotland	Yes	Listed as a Priority species in Scotland (Scottish Biodiversity List) http://www.gov.scot/Topics/Environment/Wildlife-Habitats/16118/Biodiversitylist/SBL
	No	Not listed as a Priority species in Scotland

Table 5

Definitions of criteria used to record the ecology of the oak-associated species within the OakEcol database.

Measure	Category	Definition
Age of tree used	Seed	Species uses acorns
	Seedling	Species uses the oak trees 1 year old or less
	Sapling	Species uses the oak trees under 2 m in height
	Pole	species uses the pole stage oak trees which are defined as >2 m height, younger than 50 years
	Mature	Species uses mature oak trees which are defined as more than 2 m in height and under 3 m in girth, don't have lots of holes, rotten wood etc - characteristics which are veteran tree characters
	Veteran	Species uses Veteran oak trees which includes Veteran, Ancient and Notable trees: defined as large girth (usually over 3 m, taking into consideration environmental conditions) with at least 3 veteran attributes (e.g. important habitats visible such as dead wood in the trunk, contain standing dead wood, have fallen wood around base, rot holes, water pockets, seepage lines, hollows in trunk or major limbs etc.)
Use of tree	Dead	Species uses dead oak trees
	Seed	Species uses acorns
	Feeding - direct	Species eats part of the oak (e.g. acorns, leaves, wood, bark)
	Feeding - indirect	Species eats another organism found on the oak
	Habitat - living space	Species uses the oak as a habitat in which to live (e.g. epiphytes, bird nest holes etc.)
Part of tree used	Bark	If the species grows/lives/eats/otherwise uses the bark of oak trees (used for lichens/bryophytes that grow on the bark of the tree and birds that hunt for insects on the bark of the tree. For species that nest in holes trunk was used.
	Canopy	The species grows/lives/eats/otherwise uses the canopy of oak trees
	Dead wood	The species grows/lives/eats/otherwise uses the dead wood of oak trees
	Flowers	The species grows/lives/eats/otherwise uses the flowers of oak trees
	Leaves	The species grows/lives/eats/otherwise uses the leaves of oak trees
	Limbs/branches/twigs	The species grows/lives/eats/otherwise uses the limbs/branches/twigs of oak trees
	Roots	The species grows/lives/eats/otherwise uses the roots of oak trees
	Seeds	The species grows/lives/eats/otherwise uses the seeds of oak trees
	Shoots	The species grows/lives/eats/otherwise uses the shoots of oak trees
	Trunk	The species grows/lives/eats/otherwise uses the trunk of oak trees (used for birds/mammals/inverts that live/breed inside holes in trees, but bark was used for lichens/bryophytes that grow on the bark of the tree.

Table 5 (continued)

Measure	Category	Definition
Woodland type	Woodland habitat not tree	The species uses the oak woodland habitat but is not specifically associated with oak trees (many mammals)
	Ancient woodland	Species uses oak trees in Ancient Woodland sites – any site that has always been wooded since at least 1600AD (in England and Wales) when the first maps appeared. Definition taken from Royal forestry society 2008 A glossary of tree terms http://www.rfs.org.uk/files/TreeTerms_RFS_17102011.pdf
	Recent woodland	Species uses oak trees in Recent woodland sites. Includes Woodland established since AD1600 that have regenerated and planted native woodland. Definition taken from Royal forestry society 2008 A glossary of tree terms http://www.rfs.org.uk/files/TreeTerms_RFS_17102011.pdf
	Wood pasture	Species uses oak trees in Wood pasture systems. An ancient system of land-use in which domestic animals were grazed within woodland or under widely scattered trees. The trees were often pollarded. Definition taken from Royal forestry society 2008 A glossary of tree terms http://www.rfs.org.uk/files/TreeTerms_RFS_17102011.pdf
	Plantation	Species uses oak plantations
	Non-woodland	Species uses oak trees in non-woodland situations e.g. single trees, trees in gardens, parks, hedges

Table 6

The 30 tree species for which information was collated for whether the 2300 oak-associated species will or will not use these trees in addition to oak.

Latin	English
<i>Acer campestre</i>	Field Maple
<i>Acer pseudoplatanus</i>	Sycamore
<i>Alnus glutinosa</i>	Alder
<i>Betula pendula</i>	Silver Birch
<i>Betula pubescens</i>	Downy Birch
<i>Carpinus betulus</i>	Hornbeam
<i>Castanea sativa</i>	Sweet Chestnut
<i>Fagus sylvatica</i>	Beech
<i>Fraxinus excelsior</i>	Common Ash
<i>Ilex aquifolium</i>	Holly
<i>Larix spp</i>	Larch
<i>Malus sylvestris</i>	Crab Apple
<i>Picea abies</i>	Norway Spruce
<i>Pinus nigra ssp. laricio</i>	Corsican Pine
<i>Pinus sylvestris</i>	Scots Pine
<i>Populus tremula</i>	Aspen
<i>Prunus avium</i>	Wild Cherry
<i>Pseudotsuga menziesii</i>	Douglas fir
<i>Quercus cerris</i>	Turkey oak
<i>Quercus rubra</i>	Red Oak
<i>Sorbus aria</i>	Whitebeam
<i>Sorbus aucuparia</i>	Rowan
<i>Sorbus torminalis</i>	Wild service tree
<i>Taxus baccata</i>	Yew
<i>Thuja plicata</i>	Western red cedar
<i>Tilia cordata</i>	Small leaved lime
<i>Tilia platyphyllos</i>	Large leaved lime
<i>Tilia vulgaris</i>	Hybrid T. cordata × T. platyphyllos
<i>Tsuga heterophylla</i>	Western hemlock
<i>Ulmus glabra</i>	Wych elm

Table 7
Definitions of criteria used to assess association of oak-associated species with 30 alternative tree species.

Association	Definition
No	Known not to use this tree species.
Parasite	Species is a parasite, so no assessment of alternative tree species made
Probable	Based on ecological knowledge of the species the oak-associated species is thought likely or probable to use this tree species but there are no records of the species using this particular tree species. For example, the species is known to use a wide range of deciduous tree species, thus it is probable that it will also occur on other deciduous tree species, even if no records of its occurrence on this tree species exist.
Rarely_Gen	The oak-associated species has very occasionally been recorded on this genus but there is no information on if it will occasionally use this species. Unlikely to be a good alternative tree species
Rarely_Sp	The oak-associated species has very occasionally been recorded on this tree species but very rarely, so unlikely to be a good alternative tree species
Unknown	The use (or otherwise) of this tree is unknown
Yes_Gen	Known to use this genus but no information on if it will use this species
Yes_Sp	Known to use this tree species. Note there are a few obligate species in the data base, which although only recorded on native oak in the UK have some evidence from outside the UK that they will use other tree species (often another species of oak). Hence occasionally an obligate species may have “Yes_Sp” recorded against it for an alternative tree species. Looking at the related data under the column “Association-confidence” (Table 7) where the data is shown as coming from a non-UK source explains this seeming contradiction.

Table 8
Definitions of criteria used to record the quality of the data used to make assessments of the level of association of species with either oak or any of the alternative tree species.

Data type	Definition
Anecdotal	Information on the use the species makes of oak is predominantly based on anecdotal evidence
Not reviewed-Non UK	Information on the use the species makes of oak is predominantly based on literature that has an unknown review process and uses data from outside the UK
Not reviewed-UK	Information on the use the species makes of oak is predominantly based on literature that has an unknown review process but is based on UK data
Peer reviewed-Non UK	The species is known to occur in the UK, but the information used to assess the level of association of the species with oak is predominantly based on peer reviewed literature from outside the UK
Peer reviewed-UK	Information on the use the species makes of oak is predominantly based on peer reviewed literature using data from the UK. This includes published books and quality-controlled databases.

Information on the ecology of the oak-associated species was collated. This information covered how the species used the tree (feeding directly/feeding indirectly/living space/nesting/roosting), the part of the tree used, the age of tree used, the tree form used and the woodland type used (Table 5).

The literature was searched in the same manner as that used to identify oak-associated species (Table 3) to collate evidence of whether the oak-associated species did or did not use 30 alternative tree species (see list of tree species in Table 6). The level of use of the alternative tree species was categorised according to whether the species was known to use that particular tree species, or just that genus of tree and if the species regularly used the tree species or only rarely (Table 7).

The quality of the data used to make the assessment of the level of association of the species with oak and with the 30 alternative tree species was recorded with the categories based on whether the literature was peer reviewed/the database quality controlled and if the data were from the UK or not (Table 8). In addition, all the data sources for assessments were recorded.

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Conflict of interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Past and future trends in fire weather for the UK

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Abstract. Past and future trends in the frequency of high-danger fire weather conditions have been analysed for the UK. An analysis of satellite-derived burned area data from the last 18 years has identified the seasonal cycle with a peak in spring and a secondary peak in summer, a high level of interannual variability, and a lack of a significant trend despite some large events occurring in the last few years. These results were confirmed with a longer series of fire weather indices back to 1979. The Initial Spread Index (ISI) has been used for spring, as this reflects the moisture of fine fuel surface vegetation, whereas conditions conducive to summer wildfires are hot, dry weather reflected in the moisture of deeper organic layers which is encompassed in the Fire Weather Index (FWI).

Future projections are assessed using an ensemble of regional climate models from the UK Climate Projections, combining variables to derive the fire weather indices. The results show a large increase in hazardous fire weather conditions in summer. At 2 °C global warming relative to 1850–1900, the frequency of days with “very high” fire danger is projected to double compared to the recent historical period. This frequency increases by a factor of 5 at 4 °C of global warming. Smaller increases are projected for spring, with a 150 % increase for England at 2 °C of global warming and a doubling at 4 °C. A particularly large projected increase for late summer and early autumn suggests a possible extension of the wildfire season, depending on fuel availability.

These results suggest that wildfire can be considered an “emergent risk” for the UK, as past events have not had widespread major impacts, but this could change in future, with adaptation actions being required to manage the future risk. The large increase in risk between the 2 and 4 °C levels of global warming highlights the importance of global efforts to keep warming below 2 °C.

1 Introduction

In recent years, large wildfires have occurred across many parts of the world. Areas affected have included Australia, California, Amazonia, Siberia and southern Europe. Many of these events have had severe impacts, such as loss of human lives, loss of wildlife and their habitats, destruction of houses and buildings, and degraded air quality (Deb et al., 2020; Wang et al., 2021), and this has highlighted the threat posed by this natural hazard. In the UK, impacts have so far tended to be less severe, and threats to human life are rare, but major incidents have led to significant impacts on air quality, wildlife, ecology and infrastructure (Belcher et al., 2021). Irrecoverable damage to peat soils is a particular concern as they represent a significant store of carbon, especially in Scotland (Milne and Brown, 1997). The suppression of wildfires by fire and rescue services is also costly and resource intensive.

1.1 Wildfire occurrence in the UK

In the UK, severe wildfire is an intermittent hazard as the most serious incidents are concentrated in a few dry years. Wildfire incidents are most prevalent in the spring because of the availability of dead and dry fine vegetation as fuel, but widespread wildfires have also occurred in some hot, dry summers. The UK Fire and Rescue Services attend to around 32 000 wildfires each year (Forestry Commission, 2019). However, the vast majority of wildfires are small incidents – Gazzard et al. (2016) reported that 99 % of Great Britain fires affect less than 1 ha. Most wildfires occur at the rural–urban interface or on arable land, as this is where fires are most likely to be ignited by human activity. However, many of the larger wildfires occur in more remote areas, especially moorland, forests and peatland bog. In these areas,

suppression is more difficult, and there is a greater continuity of fuel available. Forest and woodland fires currently constitute a relatively small fraction of all wildfire incidents, but their impacts can be large and costly.

For example, the Swinley Forest fire in April–May 2011 damaged 110 ha of habitat and was particularly resource intensive for the fire and rescue service, occurring close to residential areas in Berkshire, southeast England (Brown et al., 2016). Another event which occurred recently was the Wareham Forest fire, which affected 200 ha in Dorset in May 2020 (Belcher et al., 2021). As of yet, forest fires have not directly impacted residential properties in the UK, but with a number of settlements being located in or near forested land, there is a level of exposure to wildfire should fires occur in these locations. After the severe fire season in 2011, severe wildfire was added to the National Risk Register in 2013 (Gazzard et al., 2016).

1.2 Drivers of wildfire activity

Weather conditions have a direct impact on the likelihood and severity of fires occurring. Variations in weather such as temperature, relative humidity and precipitation affect the amount of moisture held in both live and dead vegetation (“fuel”) – which is crucial for flammability. As well as short-term conditions, soil moisture is affected by antecedent rainfall amounts over previous weeks and months. Fuel availability is higher in the spring and summer than in the autumn and winter. In spring, a large amount of ground vegetation is dry (not in a growth phase yet) or dead and acts as fuel. Fires in spring are generally fuelled by dead grasses, heather and surface vegetation litter. These fuels become very dry during warm weather when the relative humidity is low (50 % or less). This typically occurs when high pressure dominates the UK or the wind direction is from the east, bringing dry continental air to the UK. Wind speed also influences fuel moisture but is more closely related to the behaviour of fire, favouring spread. Strong winds help fires to spread more quickly, and there is usually insufficient green vegetation in spring to prevent the spread of a fire. In summer, however, fires are generally driven by high temperatures and prolonged dry conditions which reduce the amount of moisture in the living vegetation. Both seasons exhibit spells of dry, warm/hot, windy weather, which therefore favours the start and spread of wildfires, but conditions can fluctuate greatly on a daily basis. More generally, weather also impacts vegetation growth and thereby future fuel availability. Spatial variations in climate are also an important factor for the type and structure of vegetation.

As well as having suitable conditions to spread, wildfires need an ignition source. Lightning is a weather event that can cause wildfires to ignite, particularly in the case of “dry lightning” (not accompanied by significant rainfall) (Read et al., 2018). This is a significant source of ignitions in some parts of the world, for example Australia and Canada, but is very

rare in the UK where the vast majority of wildfires are started by human activity, either accidentally or deliberately.

In addition to the weather, the occurrence and spread of wildfire are also affected by human factors including accidental or deliberate ignition, land use, land management and any efforts to suppress the fire. Land management such as clearing of vegetation and prescribed burning can reduce fuel availability. In terms of a risk framework, these human factors can be considered to affect the vulnerability and resilience of an area to the wildfire hazard. The risk, or possibility of adverse impacts, is also affected by the value of natural and infrastructure assets exposed to the wildfire hazard.

1.3 Fire danger

Fire danger indices are a long-standing attempt at combining wildfire-related weather information into a value which represents the danger that a wildfire would pose should one be ignited. The weather variables most commonly used in the calculation of these indices are air temperature, relative humidity, wind speed and precipitation. Various such indices exist and are in use worldwide, for example the Canadian Fire Weather Index (van Wagner, 1987) and the McArthur Forest Fire Danger Index (Noble et al., 1980). Other indices add information on fuel and topography to provide a more complete picture of fire danger, such as the fire potential index which includes satellite-derived vegetation type and greenness and terrain data (Burgan et al., 1998).

This study uses the Canadian Fire Weather Index (FWI) because we focus on the impacts of climate change, which affects fire danger mainly through weather, and it is also used by the Met Office for a daily forecast service. The FWI provides a quantification of how favourable conditions are for a grass or forest fire to spread and intensify should one have been started. The model comprises six indices tracking fuel moisture content, the rate of spread and fire intensity. Calculations are based on consecutive daily observations of air temperature, relative humidity, wind speed and 24 h rainfall taken at noon, but the resulting indices are representative of the mid-afternoon-peak fire danger (van Wagner, 1987). Rainfall is also assessed over a period of several prior months to estimate the moisture content of the vegetation and soil layers. See Sect. 2.2 for further details.

1.4 Wildfire and climate change

Several studies have established links between climate change and wildfire internationally. Climate trends can increase the likelihood of severe fire weather through increases in average temperature and in the frequency, intensity, duration and/or extent of heatwaves and droughts. Fire weather seasons have significantly lengthened across 25 % of the Earth’s vegetated surface (Smith et al., 2020). The impact of anthropogenic climate change on fire weather has

emerged above natural variability for Amazonia, southern Europe/Mediterranean, Scandinavia, the western USA and Canada (Abatzoglou et al., 2019). These impacts will intensify with further warming, and future increases in fire weather risk are projected for many areas, including the Iberian Peninsula (Calheiros et al., 2021) and Canada (Wang et al., 2017). The response of lightning to climate change is uncertain, but recent research shows no significant change in lightning flash rates for the UK, with a projected decrease globally (Finney et al., 2018).

The occurrence and spread of wildfires are also influenced by human factors such as ignitions, land management and fire suppression (Wu et al., 2021). At the global scale, burned area has been found to decrease in recent decades, mainly due to agricultural expansion in savannas (Andela et al., 2017). However, in developed nations such as the UK which are already heavily urbanised, human factors are unlikely to change significantly in future, so climate is likely to remain the main factor limiting future changes (Wu et al., 2021).

Few studies have looked in detail at projected changes for the UK. Some global studies include mapped results for the UK; for example Abatzoglou et al. (2019) project climate change impacts to emerge in southern Britain at a global warming level of 2–3 °C. A review of evidence for the second UK Climate Change Risk Assessment (CCRA2) found with medium confidence that projections of drier summers with increased soil moisture deficits would suggest an increase in the number of fires and the area affected (Brown et al., 2016). They also reported greater uncertainty for changes in spring. Albertson et al. (2010) simulated local weather for the Peak District in northern England, finding a projected increase in summer wildfires after 2040. The latest set of climate projections for the UK is provided by the UK Climate Projections. The headline findings are for warmer, wetter winters and hotter, drier summers in the future (Lowe et al., 2018). Arnell et al. (2021) used 60 km resolution projections from UKCP18 and projected increases in the occurrence of high fire danger for most regions of the UK using a high-emissions scenario but very limited change with a low-emissions scenario. Warmer, wetter winters could increase fuel load due to greater vegetation growth.

1.5 Aims of this study

The wildfire hazard is represented in this study by the FWI and its sub-indices exceeding thresholds relevant for UK conditions in spring and summer. This study focuses on projected changes in the frequency of days with very high fire danger for global warming levels of 2 and 4 °C. This aims to compare possible outcomes if the world is successful in meeting targets set by the Paris Agreement in 2015 to keep the increase in global average temperature well below 2 °C above pre-industrial levels, with potential outcomes if these targets are not met. Future changes are also considered in the context of past trends in fire danger and burned area.

Section 2 of this article describes the approach taken to assess past wildfire events, the type of fire danger indices used in the analysis and how they have been calculated for past and future periods. Section 3 looks at past occurrences of moderate to severe wildfires using MODIS and EFFIS satellite-based burned areas, as well as past fire danger using the fire danger indices calculated from the ERA5 reanalysis. Section 4 looks at future projections of fire danger in a changing climate. Section 5 discusses the results and conclusions.

2 Methods and data used

2.1 Satellite-derived fire observations

Satellites are a useful tool to detect large-enough wildfires and to provide an insight into their footprint (burned area). Satellites detect wildfires through increased thermal emissions and changes in vegetation-related reflectance caused by burn scars (Humber et al., 2018). Algorithms use either or both of these to estimate the date and burned area of wildfires. Uncertainties can be considerable but are usually mitigated by the algorithms. Uncertainties typically stem from restrictions of satellite views (caused by cloud cover), spatial and temporal coverage of satellite overpasses, and non-fire-related vegetation changes such as cropland harvest or forest clearing (Boschetti et al., 2019).

Several wildfire-relevant satellite products make use of sensors of the Moderate Resolution Imaging Spectroradiometer (MODIS) on board the Terra and Aqua satellites, operated from the USA by the National Aeronautics and Space Administration. The Collection 6 product MCD64A1 (Giglio et al., 2018) provides global burned areas on a grid of $0.004^\circ \times 0.004^\circ$ of latitude and longitude – about $260\text{ m} \times 460\text{ m}$ spatial resolution for the UK – and usually has a twice-daily overpass. Validation of the product has found an omission error of 73 % and a commission error of 40 %, with approximately half the overall area mapped compared to the reference data (Boschetti et al., 2019). A total of 90 % of the area difference is due to 3 km grid cells with less than 25 % of their area burned, as small and spatially fragmented burned areas are not mapped at the 500 m scale at which the MCD64A1 product was calculated.

These data have been extracted for the UK and the total monthly burned area across the UK calculated. Data go back to 2003 and provide a 17-year-long time series of burned areas. This time series is too short for a robust assessment of trends, but it can provide useful information on monthly and inter-annual variability and an indication of recent changes. In addition to the MODIS burned area, the burned area dataset of the European Forest Fire Information System (EFFIS) was used, as a point of comparison. This dataset is also derived from the daily processing of MODIS satellite imagery. It is available from 2008 onwards and rep-

Table 1. Fire danger classes used by the Met Office for the UK (in particular England and Wales).

Level of fire danger	FFMC	ISI	BUI	FWI
1 (low)	< 63	< 2	< 20	< 4.54
2 (moderate)	63–84	2–5	20–30	4.54–9.38
3 (high)	84–88	5–10	30–40	9.38–17.35
4 (very high)	88–91	10–15	40–60	17.35–52.36
5 (exceptional)	> 91	> 15	> 60	> 52.36

resents 75 % to 80 % of the total burned area in Europe by fires larger than 30 ha in size.

2.2 Fire Weather Index

The Met Office provides a weather-related fire danger forecast for emergency planners in England and Wales. During the implementation of the system, several models were examined, and the choice was made to use the Canadian FWI system for modelling fire danger in the UK because it was found suitable to pick out periods of greater fire danger in wet and hot, dry summers as well as in spring (Met Office, 2005). It has also been in use since 2007 in the European service EFFIS.

The FWI is comprised of five sub-indices. The Initial Spread Index (ISI) is a numeric rating of the expected rate of fire spread. It combines the effects of wind and fine fuel moisture (the Fine Fuel Moisture Code, FFMC) on rate of spread without the influence of variable quantities of fuel. The Buildup Index (BUI) is a numeric rating of the amount of fuel available for combustion, which combines temperature, rainfall and relative humidity to represent the moisture content of organic layers – the Duff Moisture Code (DMC) for moderate depth layers and the Drought Code (DC) for deep layers. The ISI and BUI are then combined to arrive at the overall FWI.

In order to provide a fire danger rating, the FWI is divided into five classes ranging from 1 (low danger) to 5 (exceptional danger). The thresholds defining each class (or level of fire danger) were adapted for the UK (Kitchen, 2010), considering the frequency of values historically experienced across a selection of sites, and following the method of van Wagner (1987) based on progressive values of a fire intensity scale related to the FWI. The thresholds are shown in Table 1. The UK “exceptional danger” class was specifically designed to be very rare and is used to allow closure of public-access land. These levels are applied uniformly across the whole of the UK, which might not be reflective of the actual fire risk in each part of the country. For example, lower thresholds may be more appropriate for conditions in Scotland, with fires able to ignite above an ISI of 2. Vulnerability to wildfire is also likely to vary considerably across the country due to factors such as land use and vegetation type. Note that EFFIS

uses alternative thresholds more suitable for southern European conditions.

De Jong et al. (2016) evaluated each of the FWI components during UK wildfire events, finding that the FFMC and ISI sub-indices by themselves performed best in spring and that the overall FWI showed the greatest skill in summer. This is because most spring wildfires require dry fine fuels to allow self-sustaining ignitions, as the moisture content of vegetation is generally lower than in summer due to limited growth. In summer however, most wildfires tend to occur during prolonged dry periods, so the BUI is also important, leading to the overall FWI performing best. Similar results were obtained by Davies and Legg (2016) for Scotland, except that here the cooler, wetter climate means that drought conditions are unusual, which limits the effectiveness of the FWI in summer. The Natural Hazards Partnership Daily Hazard Assessment (Hemingway and Gunawan, 2018), an overview of natural hazards that could affect the UK over the next 5 d which is provided to emergency planners and responders, uses forecast exceedances of the ISI “high” danger level to warn of elevated fire weather conditions from November to May and exceedances of the FWI “very high” danger level from June to October. This approach has also been taken for the results presented in this report.

2.3 Historical weather data

Historical daily values of the FWI and its sub-indices have been calculated by the Copernicus Emergency Management Service using ERA5 reanalysis data (Vitolo et al., 2020). This readily available dataset was extracted from the Climate Data Store for the period 1979 to 2020 (Copernicus, 2019) and was used to provide a good proxy for observations of past fire weather conditions. Gridded reanalysis datasets provide a spatially complete and consistent record of the global atmospheric circulation, incorporating observed data from a range of sources. ERA5 is the latest generation reanalysis dataset produced by the European Centre for Medium-Range Weather Forecasts (ECMWF) (Hersbach et al., 2020). Its spatial resolution is 0.25° in latitude and longitude (approximately 28 km × 15 km over the UK).

2.4 UKCP18 climate model scenarios

Future scenarios of climate change are based on outputs from one of the components of the UK Climate Projections 2018 (UKCP18). This component uses an up-to-date climate model downscaled to a 12 km scale over the UK to assess how the climate may change in response to increasing radiative forcing (Met Office Hadley Centre, 2018). These projections comprise a perturbed parameter ensemble of 12 members (PPE-12) of the Met Office Hadley Centre regional climate model HadREM3-GA705 (Fung et al., 2018), providing a range of plausible outcomes that can reveal the potential for extreme conditions. However, these regional projections

are downscaled from a perturbed-parameter ensemble of the Met Office Hadley Centre global climate model HadGEM3-GC3.05 (Fung et al., 2018), and as such they do not cover the full range of uncertainty. In addition, HadGEM3-GC3.05 samples the warmer end of the warming climate response – the model has a higher climate sensitivity to greenhouse gases (Lowe et al., 2018).

We examine projected changes in fire weather global warming levels (GWLs) of 2 and 4 °C above 1850–1900, since these levels of warming are routinely examined in the UK Climate Change Risk Assessment (Committee on Climate Change, 2017). Global warming of 4 °C by 2100 is within the range of possible outcomes for emissions scenarios consistent with a continuation of current policies (Betts, 2020). We obtain projected regional climate changes at these GWLs from projections driven by the RCP8.5 emissions scenario (Murphy et al., 2018). This is a scenario of very high emissions, above those considered consistent with a continuation of current worldwide policies and is used to produce high-end climate change scenarios for use in risk assessments – it is not intended to be interpreted as a projection of the most likely outcomes. Projections with this scenario reach GWLs of 2 and 4 °C earlier than would be reached with emissions scenarios consistent with current policies. However, for quantities known to scale linearly with global warming levels, such as many aspects of extreme weather (Wartenburger et al., 2017), it can be appropriate to treat such changes as representative of the regional climate state reached at the same level of warming at later times (Bärring and Strandberg, 2018). Periods corresponding to global warming levels of 2 and 4 °C above 1850–1900 were identified in the RCP8.5 12 km projections and are used as a proxy for the future wildfire hazard at these levels, which may then be translated to other pathways of greenhouse-gas emissions. For each ensemble member, an 11-year period centred on the year when the model run first reached the relevant global warming level was used, as shown in Table 2 of the UKCP18 report on derived projections (Gohar et al., 2018). For the 2 °C warming level, these central years range from 2027 to 2035 and for the 4 °C level from 2057 to 2070 across the ensemble.

Modelled past and future fire danger was calculated with a Python 3 implementation of the Canadian FWI system (Wang et al., 2015) with the relevant fire danger classes for application to the UK (Table 1). Input weather variables from the regional UKCP18 datasets were daily values of maximum air temperature, mean relative humidity, mean wind speed and precipitation. These were used as a proxy for noon values, with precipitation totals from noon to noon being estimated by averaging two daily values. The use of daily values over actual noon values likely introduces a slight bias in the results, but this is mitigated by using the daily maximum temperature over the daily mean temperature, as the former is usually close to the noon value.

All climate models exhibit differences between modelled results and observations. Calibration techniques, such as bias

adjustment, can be used to account for systematic differences. However, bias adjustment of multivariate indices such as the FWI is complex, and there are issues both with applying techniques directly to the index and with separately adjusting each individual weather parameter (Casanueva et al., 2018). Therefore, calibration techniques have not been applied to the results, and the model-derived results shown should be interpreted with care. While future trends such as an increasing occurrence of severe fire danger are likely to be robust, absolute numbers such as percentages of days exceeding certain threshold values would be likely to change if a recalibration technique is applied to the model data.

3 Past trends in UK wildfire occurrence

There are no long-enough records on the occurrence of wildfires in the UK to allow for the estimation of trends. Wildfire incidents in England have been recorded in the Fire & Rescue Services Incident Recording System since 2009 only. Therefore, past trends have been assessed using satellite-derived data on burned area and fire weather indices.

3.1 Burned area

The monthly UK burned areas recorded by the MODIS satellite from 2003 to 2019 are shown in Fig. 1, and the corresponding annual cycle of the multiple-year mean burned area for each month of the year is shown in Fig. 2. These show that the greatest total burned area has occurred in spring, with a peak in April. There is also a significant total for February, mainly due to events in 2003, although there have also been February wildfires in recent years (BBC News, 2019a). The annual cycle shows a secondary peak in the summer around the month of July. The three largest multi-month burned areas detected by the MODIS satellite over the last 17 years are indeed from the springs of 2019, 2003 and 2007. These springs were all characterised by warm and dry conditions conducive to wildfires, leading to events which had significant impacts. In the spring of 2019, Scotland was hit by several wildfires, two of which were particularly large (BBC News, 2019b). The fire in Moray in April burned approximately 50 km² of grassland and threatened a wind farm. The fire in Sutherland in May burned 80 km² of moorland (heather, grass and dry peatland) including part of the Forsinard Flows RSPB natural reserve and threatened electricity supply to 800 homes. In 2003, several fires occurred in the Peak District in April; the Bleaklow peat fire, the most severe, burned for a week and destroyed about 7 km² of moorland including protected wildlife areas. Its smoke also led to the closure of major roads and disrupted air traffic at the airport of Manchester, and there were economic costs associated with restoration and suppression. The large number of spring fires was associated with a lack of winter snow cover followed by a period of cold air and strong sunshine, which

prolonged the period during which grounds were frozen, causing significant damage to plants (Davies, 2008).

The greatest summer burned areas occurred in July 2006 and July 2018 (Fig. 1). Both months were warmer, sunnier and drier than the average and saw several major wildfire events. Notably in 2018, the Saddleworth Moor and Winter Hill fires started in late June and burned for over 3 weeks, threatening properties and vital infrastructure, and damaged about 18 km² of moorland (Sibley, 2019). Other fires also occurred on defence ranges, e.g. on Salisbury Plain. The summer of 2018 was hot and dry and was preceded by cold periods in March and early April 2018, thought to have delayed the start of the growing season.

The MODIS burned areas, as annual totals, are shown in Fig. 3 along with the EFFIS burned areas (available from 2008 onwards). The greatest burned areas are confirmed for 2019, 2003, 2007, 2018 and 2011. There is a good correlation between the two sources (Pearson's $r = 0.82$), but the EFFIS burned area values tend to be higher than MODIS values. They have been produced using different processing methods that may allow more moderately sized wildfires to be included. Overall, no statistically significant trend was found in the MODIS data from 2003 (Mann–Kendall nonparametric trend test). A significant positive trend was identified in the EFFIS data, but little reliance can be placed on this result due to the short duration of the series.

3.2 Fire weather

Although they do not represent actual wildfire events, the fire danger indices calculated from the ERA5 reanalysis constitute a long record (1979 to date) over which to assess trend and variability of the potential for severe wildfire seasons. Spring and summer wildfire seasons are analysed separately using different fire danger indices, as they are clearly distinct in behaviour and origins. Spring conditions are investigated using the ISI, summer conditions using the FWI (see Sect. 2.2 for further detail).

For each season, the percentage of days with dangerous fire weather conditions was found for each grid box, and these percentages were then averaged across the UK. In summer, this means any days in June, July or August with very high FWI fire danger ($\text{FWI} > 17.35$); in spring, this means any days in March, April and May with high ISI fire danger ($\text{ISI} > 5$). Figure 4 shows the time series from 1979 to 2020 for spring, and Fig. 5 shows the time series for summer. In spring, the most frequent occurrence of high fire danger was clearly in the most recent year of 2020, with 2011 having the next highest frequency. These were the two warmest springs in over 100 years for the UK and were also much drier than average (Met Office, 2021). These years had a large number of wildfire events during spring, although the burned area was not as great as for 2019, which was a more average year in terms of the overall frequency of high fire danger conditions.

A non-parametric Mann–Kendall test indicated that there was no statistically significant trend, pointing to weather-related fire conditions in spring having not changed significantly over the past 40 years despite the most recent year having the highest frequency of high fire danger. The more episodic nature of wildfires in summer is clearly visible when comparing Fig. 5 to Fig. 4. The severe seasons of 1990, 1995 and 2018 (and 2006) stand out, but most years are characterised by a much smaller occurrence of very high fire danger. A Mann–Kendall test again found no evidence of a trend.

Months with very high fire danger in summer were found to be closely related to the MODIS burned area (Pearson correlation coefficient of 0.7 for June and July). For spring, there is a moderate positive relationship between burned area and fire weather indices, with monthly correlation coefficients ranging from 0.3 to 0.6. While severe wildfire seasons generally occur when the fire danger is higher than normal, high fire danger does not automatically imply the occurrence of wildfires because fire danger indices only assess how difficult a wildfire would be to control should one be ignited. In other words, if the fire danger is severe, fires, if they start, are likely to have a much bigger impact.

4 Future projections of fire weather

4.1 Fire danger levels

Figures 6 and 7 summarise the modelled levels of fire danger in spring and summer for England, Northern Ireland, Scotland and Wales, during the historical period and in a world with 2 and 4 °C of warming. The plots are based on the daily 95th percentile index values, meaning that 5 % of the area of the country is affected by higher danger levels. As before, the ISI is used to represent fire weather conditions in spring and the FWI in summer, using the danger class thresholds shown in Table 1. However, the “exceptional” FWI class ($\text{FWI} > 52.36$) has been encompassed into a broader “extreme” class start at $\text{FWI} > 38$, as the exceptional danger level is – by design – very rarely reached in the data.

In spring, a slight increase in moderate and high levels of modelled fire danger is projected for each of the UK countries (Fig. 6). There is a corresponding slight projected reduction in the frequency of low fire danger. The change is most marked in England, where the frequency of high fire danger covering at least 5 % of the country is projected to increase from 6 % of days for the historical period of 1981–2010 to 9 % at the 2 °C global warming level and 12 % at the 4 °C level.

In summer, modelled fire danger increases markedly between the reference period and the 2 °C global warming level, with large increases projected for the 4 °C level (Fig. 7). In England and Wales, the percentage of days with at least 5 % of the country at the very high or extreme danger levels is projected to double between the reference period and

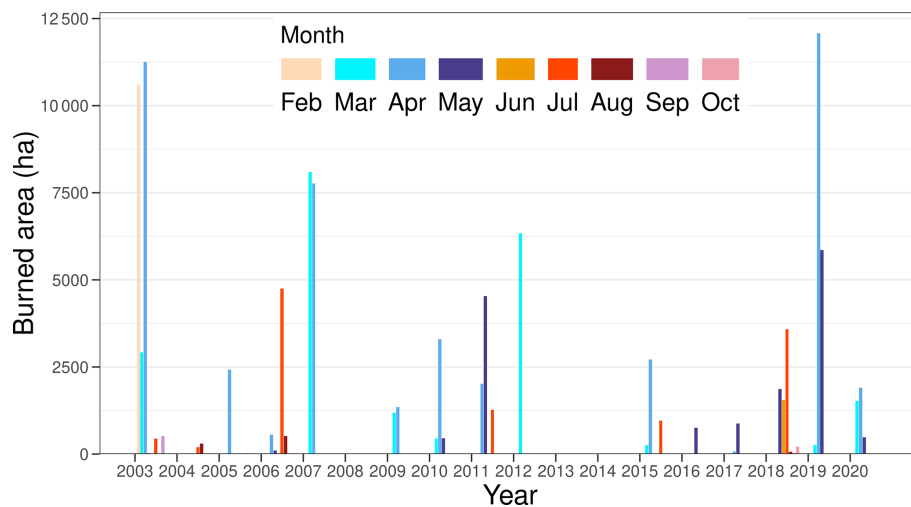


Figure 1. Monthly burned area (km^2) for the UK from 2003 to 2020 from MODIS MCD64A1 data.

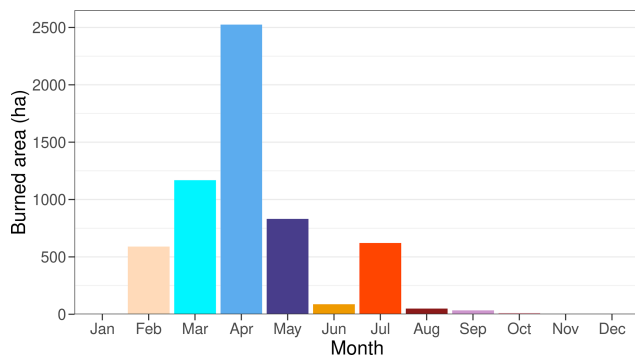


Figure 2. Monthly mean burned area (km^2) for the UK from 2003 to 2020 (from MODIS MCD64A1 data product).

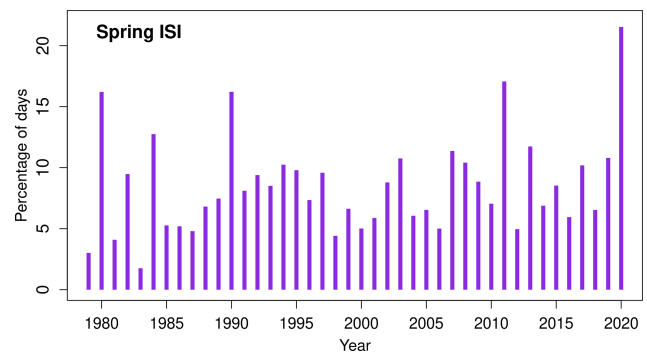


Figure 4. Annual percentage of spring (March–April–May) days, 1979–2020, averaged over the UK, with high fire danger ($\text{ISI} > 5$), based on ERA5 reanalysis data.

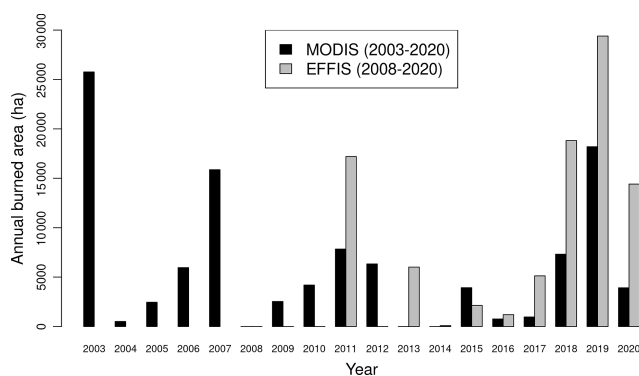


Figure 3. Annual burned area (km^2) for the UK, estimated from MODIS MCD64A1 (2003–2020) and EFFIS (2008–2020).

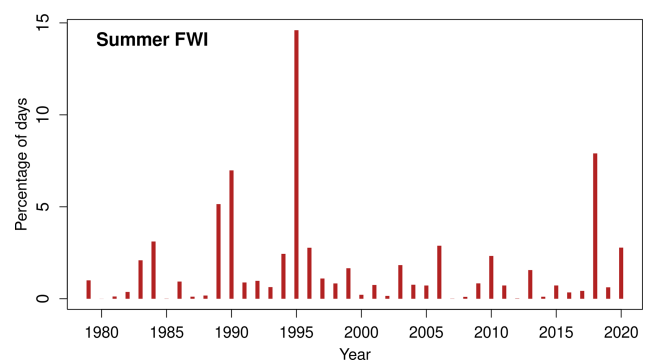


Figure 5. Annual percentage of summer (June–July–August) days, 1979–2020, averaged over the UK, with very high fire danger ($\text{FWI} > 17.35$), based on ERA5 reanalysis data.

the 2°C global warming level and to increase by five times the reference frequency at the 4°C level, when the frequency reaches 46 % of summer days for England. In Scotland and Northern Ireland, there is a large projected increase in the

frequency of high fire danger. Very high danger levels remain rare but are projected to start to occur more frequently over at least 5 % of the country by the 4°C level. The increase in

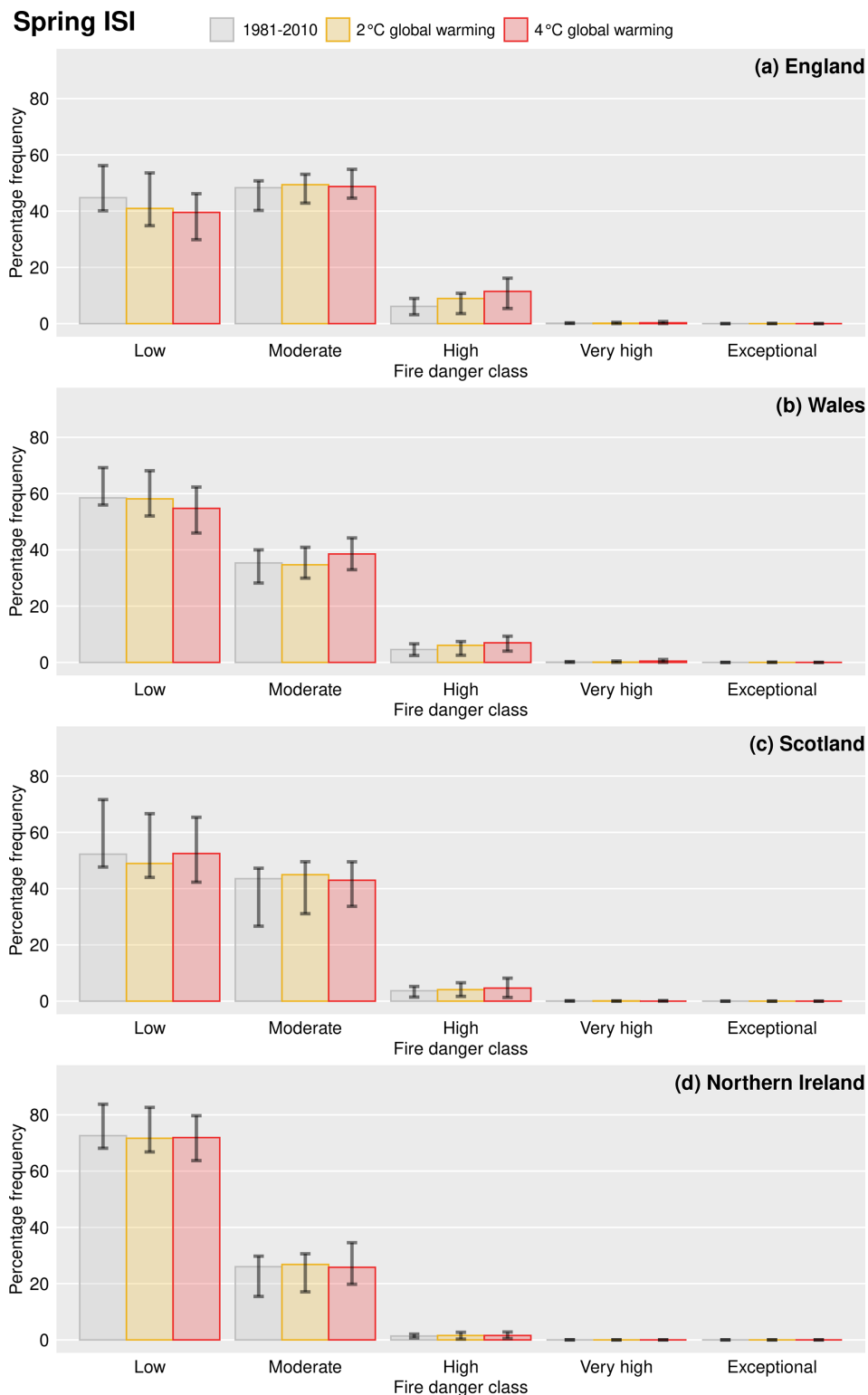


Figure 6. Modelled percentage frequency of spring fire danger classes based on the daily 95th percentile ISI across each country – (a) England, (b) Wales, (c) Scotland and (d) Northern Ireland. Historical period (1981–2010), 2 and 4 °C global warming levels. Coloured bars show the median of the 12 ensemble members, and the error bars show the 10th–90th percentile range.

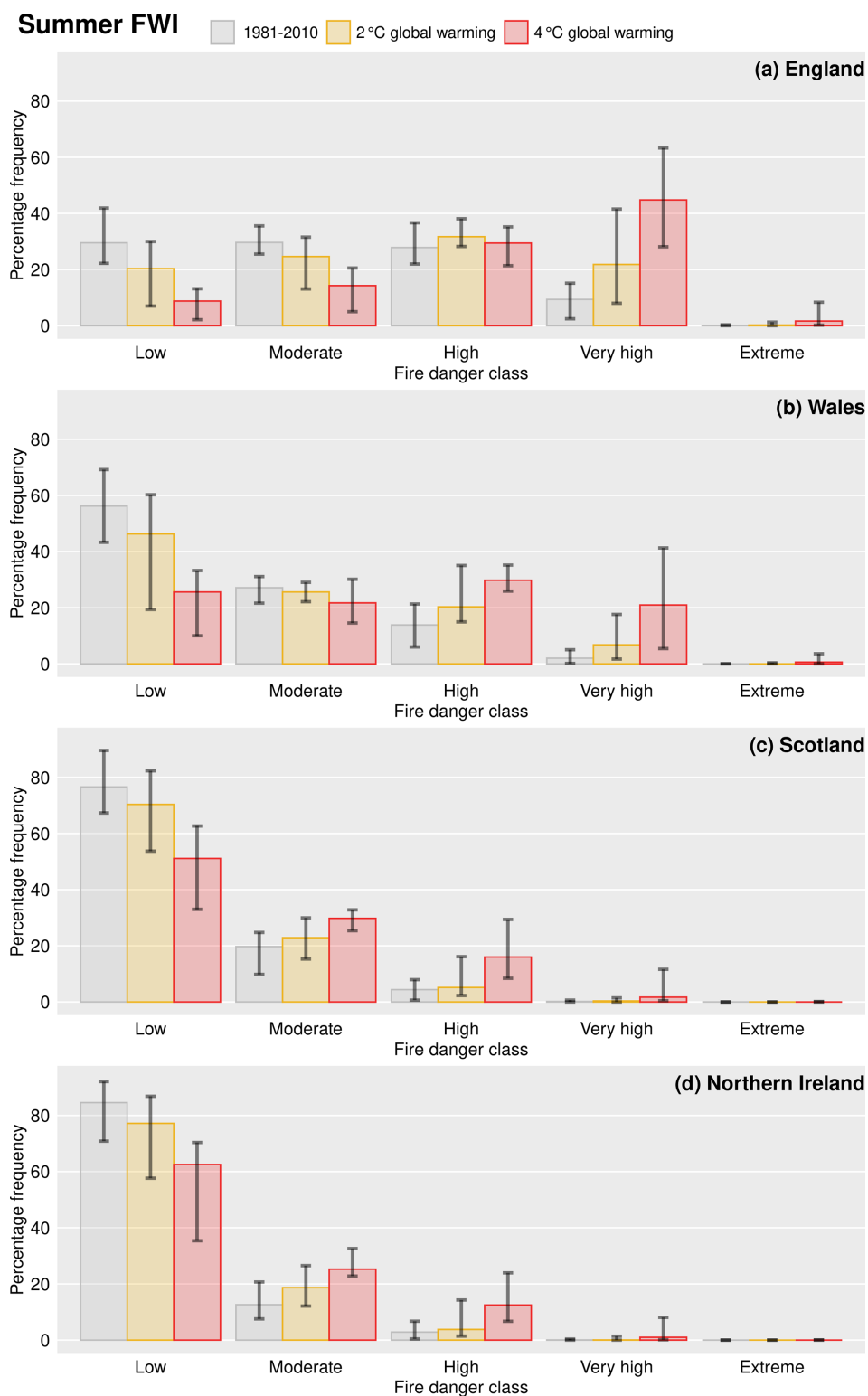


Figure 7. As Fig. 6, but for summer fire danger classes based on the daily 95th percentile FWI. The exceptional danger level has been encompassed into a broader extreme danger level with FWI > 38.

the percentage frequency of the high or very high levels is less than 2-fold for the 2 °C level and approximately 4-fold for the 4 °C level.

4.2 Monthly changes

Table 2 shows the results broken down by month from February to October, extending the previous results for spring and summer to include the winter month of February and the autumn months of September and October. The modelled percentage of days with very high FWI across at least 5 % of the UK is projected to at least double for the 2 °C global warming level for all months from June to October and to increase by at least a factor of 5 by the 4 °C global warming level for these months. Particularly large increases are projected for September, which could indicate an extension of the wildfire season into autumn, a season when previously few wildfires have occurred in the UK, although this is dependent on fuel availability. The ISI is a more useful index for the spring and late winter, and Table 2 shows no increase in the frequency of high fire danger in February. Slight increases are shown for March and April, with a larger increase for May.

These results highlight the importance of meeting the Paris Agreement targets, as there is a large difference between the fire risk in the projections for a 2 °C world compared to a 4 °C world. Even in a 2 °C world, however, the results show a marked increase in dangerous fire weather in some areas, highlighting the benefits of taking action which would keep global warming below 2 °C.

4.3 Spatial changes

Spatially, the frequency of severe fire weather is not uniform across the UK. Figure 8 shows the percentage of days across the UK with dangerous fire weather in spring or summer, based on the historical ERA5 reanalysis data. In spring, high fire danger levels based on the ISI are most frequent over eastern, central and southern England. Elevated frequencies also found for coastal parts of northern England, Scotland and Wales due to higher wind speeds, especially for more exposed coastlines. In summer, dangerous fire weather is found most frequently for central and southern England. However, a bias is visible between the ERA5 data and the UKCP18 climate model data for the historical period shown in Figs. 8a and 9a, with fewer days of high fire danger in the latter for the spring. The bias is much lower for the summer FWI, comparing Figs. 8b and 10a.

Projections for the future point towards a gradual strengthening of severe fire conditions in the same regions rather than a drastic change in geographical patterns. In summer, a stronger increase is visible in Fig. 10, with values across much of England ranging from 5 % to 20 % in a 2 °C world and from 15 % to 40 % in a 4 °C world. The projected changes are particularly dramatic in central and southern England and south Wales, and especially for the 4 °C global

warming level, again emphasising the benefits of keeping global warming below 2 °C. The highest frequencies for each time period are found in southeast England.

The number of days with severe fire danger is projected to remain relatively low in northern England and Scotland, but still with marked projected increases for the future periods. However, many of the most significant historical wildfire events have taken place in these northern regions. It is likely that these areas are more vulnerable to wildfire due to their more rural nature and increased remoteness, which leads to greater fuel availability and increases the risk of fires being able to spread before they can be suppressed. The spatial distribution of different types of vegetation fuels also affects vulnerability to levels of the different fire weather indices (de Jong et al., 2016).

4.4 Weather drivers

Table 3 shows changes in the four weather variables which contribute to the calculation of the fire weather index, using the same methods as for changes in the fire weather indices (see Sect. 2.4). Increases in daily maximum temperature and decreases in relative humidity are higher in summer than in spring, and at the 4 °C global warming level compared to the 2 °C level. For precipitation, an extra month before the season considered has been included to account for the antecedent rainfall component of the indices. Precipitation is projected to increase slightly from February to May but to decrease from May to August. Slight decreases in wind speed are projected, again with a greater change shown for summer.

Changes in the weather variables were compared to the changes in the fire weather indices shown in Figs. 6 and 7, for individual ensemble members, by country and period (the 2 and 4 °C global warming levels compared to the 1981–2010 period). This revealed a strong negative relationship between change in relative humidity and change in the percentage of days with very high fire weather index in summer (see Fig. 11). The R^2 value from a linear model including only relative humidity is 0.76, compared with 0.84 when including all four weather variables, and no more than 0.49 when relative humidity is omitted as a coefficient (see Table A1). Summer increases in temperature and decreases in precipitation are both correlated with decreases in relative humidity, and both also have an effect on the projected increases in dangerous fire weather, but decreasing relative humidity appears to have the biggest influence. The factors influencing the smaller changes in fire weather projected for spring are less clear, with an R^2 of 0.53, with relative humidity again the most significant coefficient, followed by temperature (see Table A1). The slight projected increase in precipitation and the lower changes in temperature and relative humidity compared to summer are likely to explain the lower projected increases in weather-related fire danger in spring.

Willett et al. (2020) showed that relative humidity has decreased over recent decades for mid-latitude oceanic re-

Table 2. Spatial 95th percentile across the UK of the modelled monthly percentage of days with dangerous fire weather (high ISI (> 5) from February to May, very high FWI (> 17.35) from June to October). Median and (in brackets) 10th–90th percentile uncertainty range from the 12 ensemble members. Values are quoted to two significant figures.

Month	1981–2010	2 °C global warming	4 °C global warming
February	1.8 (0.2–2.9)	1.5 (0.0–4.4)	1.5 (0.0–3.0)
March	2.8 (0.5–4.2)	3.0 (0.9–6.5)	3.0 (0.9–7.5)
April	5.8 (3.0–8.4)	6.4 (2.5–11)	8.2 (1.6–13)
May	8.4 (3.4–15)	14 (4.4–20)	18 (9.8–30)
June	3.3 (0.3–6.0)	8.0 (1.2–24)	23 (7.0–34)
July	9.2 (2.4–17)	27 (6.3–44)	44 (21–76)
August	9.1 (1.8–19)	25 (7.4–52)	55 (36–86)
September	3.2 (0.3–10)	18 (5.5–42)	44 (19–69)
October	0.2 (0.0–2.0)	1.1 (0.3–2.7)	6.4 (1.9–18)

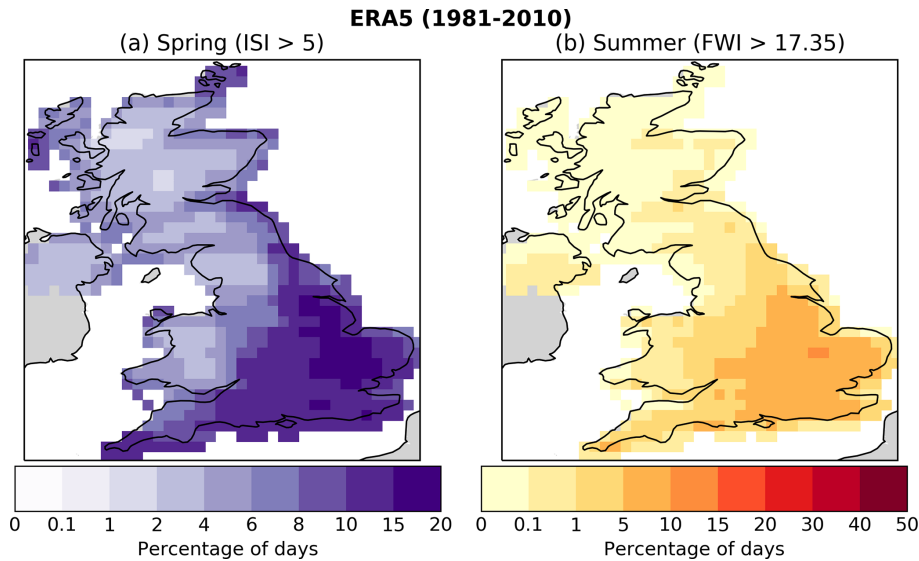


Figure 8. Percentage of days with dangerous fire weather for (a) spring (ISI > 5; at least high) and (b) summer (FWI > 17.35; at least very high). Based on the ERA5 reanalysis data for the 1981–2010 period.

gions including around the UK. Warmer temperatures and lower relative humidity increase evaporative demand in the atmosphere, which increases evapotranspiration, depletes soil moisture and increases plant water stress, and this can lead to agricultural and ecological drought (Douville et al., 2021). Agricultural and ecological drought is associated with favourable conditions for fire spread as represented in elevated values of the FWI, especially in summer for the UK.

5 Discussion

Historical variability and trend in the occurrence of wild-fires in the UK have been assessed using satellite-derived burned area data over the past 17 years (2003 to 2019) and reanalysis-derived fire danger indices over the past 40 years (1979 to 2019). In the UK, the main fire season is the spring, with the month of April having the greatest monthly total

burned area across the record. This is due to a combination of a greater availability of fuel (dry and/or dead vegetation) and conducive weather conditions (warmer, drier weather). Summer wildfires have been more episodic in nature, with most severe wildfire events being concentrated in a few hot, dry summers. No trend in historical burned area could be established, even though the past 3 years saw major wildfire incidents, and looking at historical wildfire-prone weather conditions, no significant trend could be identified either. The lack of a clear trend in summer may be due to the influence of rainfall (one of the weather variables used in the calculation of the fire danger indices), which has high interannual variability and has seen a slightly increasing trend over the UK in recent decades (Met Office, 2021). The increasing trend in temperature may have been balanced by this change in rainfall. Regions most affected by high fire danger in spring are most of central and southern England, coastal parts of north-

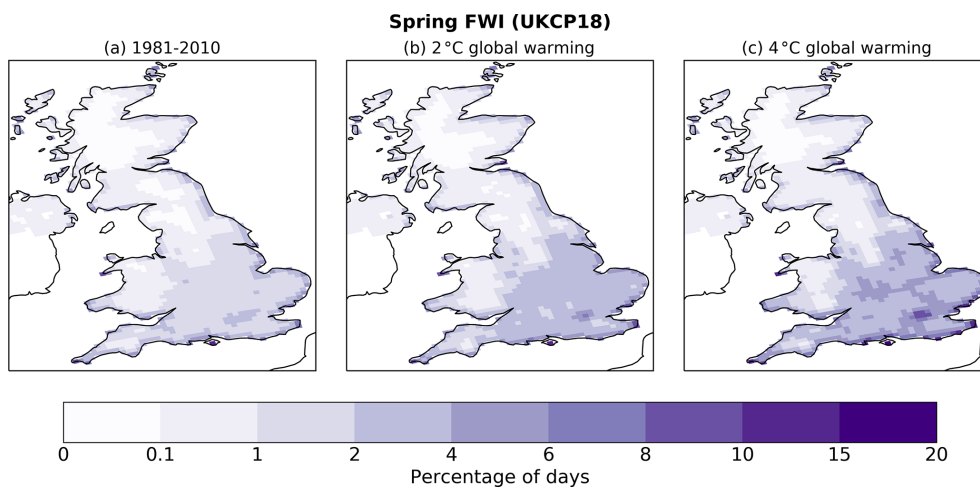


Figure 9. Percentage of spring days with high fire danger (ISI > 5) over the UK, based on the UKCP18 12 km data averaged over the 12 ensemble members. (a) Historical period (1981–2010), (b) 2 °C global warming and (c) 4 °C global warming. The colour bar is the same as in Fig. 8a.

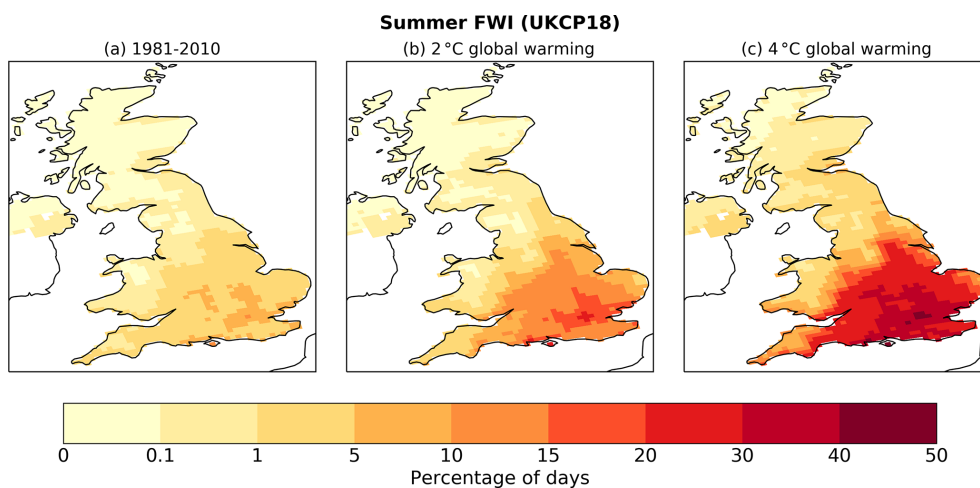


Figure 10. Percentage of summer days with very high fire danger (FWI > 17.35) over the UK, based on the UKCP18 12 km data averaged over the 12 ensemble members. (a) Historical period (1981–2010), (b) 2 °C global warming and (c) 4 °C global warming. The colour bar is the same as in Fig. 8b.

ern England, Scotland, and Wales; in summer, hazardous fire weather is mainly concentrated in most of central and southern England.

The future projections presented here only provide a partial, weather-based view of the current and future risk of severe fire danger in the UK. Indeed, the fire danger indices used are calculated purely from weather variables (temperature, relative humidity, wind and precipitation) and do not account for other factors such as ignition, vegetation abundance, vegetation type, land management or fire suppression. The UK population continues to grow gradually, and this could lead to an increase in ignitions, especially as climate change may lead to conditions which are more frequently suitable for outdoor activities (Belcher et al., 2021). Large

climate-driven changes in land use are not expected in the UK, as land cover is dominated by direct human influence. A few studies have attempted to model future changes in wildfire due to socioeconomic and vegetation effects as well as climate at a global or continental scale, but variations in their results suggest that future changes in socioeconomic and vegetation effects on wildfire are very uncertain. However, the models agree that climate is expected to be the main driver of changing risk for northern Europe (Wu et al., 2015; Wu et al., 2021). One model also showed a significant effect of increased atmospheric CO₂ concentrations on increased fuel load and thus burned area (Wu et al., 2015). Indeed, important feedbacks link a changing climate, fuels and fire regimes (Davies et al., 2008). The warmer, wetter winters

Table 3. Weather variables averaged over the UK for (a) spring and (b) summer; historical values for the 1981–2010 period and absolute changes from 1981–2010 to the 2 and 4 °C global warming levels, showing the mean and range of 12 ensemble members from the UKCP18 12 km model.

Weather variable	1981–2010	2 °C GWL change	4 °C GWL change
(a) Spring			
Daily maximum temperature (°C)	9.6	1.2 (0.6 to 1.7)	2.5 (1.8 to 3.3)
Relative humidity (%)	80.2	−0.9 (−1.8 to −0.1)	−1.4 (−2.8 to −0.7)
Surface wind speed (m s ^{−1})	4.6	0.0 (−0.2 to 0.2)	0.0 (−0.3 to 0.1)
Precipitation (mm d ^{−1})	3.5	0.1 (−0.2 to 0.3)	0.2 (−0.1 to 0.6)
(b) Summer			
Daily maximum temperature (°C)	17.6	1.9 (0.8 to 2.8)	4.4 (3.3 to 5.5)
Relative humidity (%)	78.6	−1.8 (−3.5 to −0.8)	−4.2 (−6.6 to −2.4)
Surface wind speed (m s ^{−1})	3.9	−0.1 (−0.2 to 0.1)	−0.2 (−0.3 to −0.1)
Precipitation (mm d ^{−1})	2.9	−0.1 (−0.5 to 0.0)	−0.5 (−1.0 to −0.3)

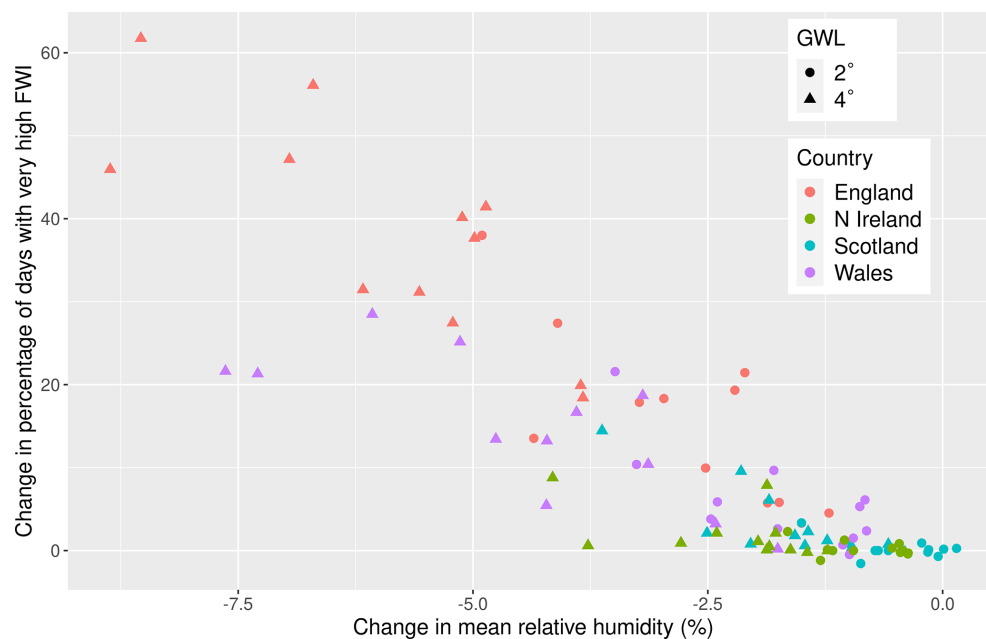


Figure 11. Scatter plot of change in the percentage of summer days with very high FWI against change in mean relative humidity (%), by country and ensemble member. Changes are from 1981–2010 to the 2 and 4 °C global warming levels.

projected for the UK may lead to increased production of vegetation and therefore increased fuel availability when allowed to dry during periods of elevated fire danger in spring. In summer, increasing periods of hot, dry weather may limit vegetation growth. In terms of spatial patterns, further research is needed on variations in socioeconomic and vegetation vulnerability to move from hazard mapping to risk mapping.

Our results show a projected doubling of the frequency of very high fire danger levels in summer for England and Wales at the 2 °C warming level and a 5-fold increase at 4 °C of global warming. Smaller increases are projected for spring,

with a 150 % increase in high fire danger for England at 2 °C of global warming and a doubling at 4 °C. These results have important policy implications for both mitigation and adaptation. Belcher et al. (2021) provide a detailed assessment of adaptation responses. Projected changes to wildfire risk will need to be recognised and incorporated into land management and design plans for a range of land uses, including new developments at the rural–urban interface and potential re-wilding, afforestation or peatland restoration schemes aimed at helping to achieve net-zero carbon emissions. In fire-prone environments, an increased capacity to manage fuel loads through both prescribed burning and mechanical

fuel removal could help to reduce the danger posed by increasingly extreme fire weather conditions. There is also a need for Fire and Rescue Services to plan for long-term increases in the training and resources required to suppress wildfires. In order to reduce ignitions, increased social understanding of wildfire is required, and public communication of periods of high fire risk is recommended through an appropriate fire danger rating system. Lessons can be learned from the example of countries like Australia, where severe events have led to the loss of life and property, leading to the development of a new fire danger rating system to support better decision-making from emergency responders and public communication of risk and actions required (Matthews et al., 2019).

The particular set of 12 km regional projections analysed does not cover the full range of uncertainty and samples the warmer end of the climate response to greenhouse gases. We examined projected changes in fire weather at the global warming levels of 2 and 4 °C, driven by the very high emissions scenario RCP8.5. Although projections with this scenario reach GWLs of 2 and 4 °C earlier than would be reached with emissions consistent with current policies, the use of GWLs allows such changes to be considered representative of the regional climate state reached at the same level of warming but translated to a later time period.

The UKCP18 model data have not been recalibrated (e.g. using bias-adjustment techniques) in this study. Comparison of the historical occurrence of severe fire danger from the ERA5 reanalysis (a good proxy for observations) with that from the climate model ensemble shows that biases are present, especially for spring. The large biases seen in spring mean that results for this season in particular should be interpreted with caution. However, future trends such as an increasing occurrence of severe fire danger are likely to be robust.

6 Conclusions

Our results suggest that wildfire can be considered an emergent risk for the UK, as past events have not had widespread major impacts, but this could change in the future.

The distribution of fire danger levels in summer revealed an increase in the frequency of very high fire danger levels being exceeded for England and Wales in a 2 °C world but a particularly dramatic increase in a 4 °C world. For Scotland and Northern Ireland, large projected increases in moderate to high fire danger were shown. A corresponding decrease in occurrence of days with low fire danger across the UK is projected. Spatially, the increase in the occurrence of severe fire weather is more pronounced in central and southern England, and in Wales.

Overall, while the occurrence of high fire danger levels in spring is not projected to change as much throughout the 21st century, the spring wildfire season is already the most

frequently severe season of the historical period. Combined with a projected increased fire danger in summer, and possibly autumn, the UK could see overall more sustained severe fire danger throughout the year. This is likely to require considerable adaptation planning and policy measures, such as increased resources for fire and rescue services, and management of land to reduce the risk of ignitions and to reduce fuel availability in high-risk areas.

The large increase in wildfire hazard between the 2 and 4 °C global warming levels emphasises the importance of meeting the targets set by the Paris Agreement in order to keep global warming levels below 2 °C and so to avoid the worst impacts of the increased risk of more frequent and intense wildfires in the UK and an extended season of hazardous fire weather conditions.

Appendix A

Table A1. Multiple R^2 values for linear models explaining changes in the percentage of days with high fire danger by ensemble member, country and future period with changes in different combinations of the weather variables mean relative humidity (RH), precipitation amount (pr), daily maximum temperature (tx) and surface wind speed (wind) in spring and summer.

Weather variables	R^2 (spring)	R^2 (summer)
All	0.53	0.84
RH + pr + tx	0.52	0.83
RH + pr + wind	0.49	0.84
RH + tx + wind	0.53	0.80
RH + pr	0.49	0.83
RH + tx	0.52	0.78
RH + wind	0.46	0.80
RH	0.46	0.76
tx + pr + wind	0.32	0.49
tx + pr	0.22	0.40
tx + wind	0.23	0.49
pr + wind	0.02	0.24
pr	0.00	0.21
tx	0.19	0.39
wind	0.01	0.02

Code and data availability. The results have been processed from three main publicly available data sources: MODIS MCD64A1 satellite-derived burned area (Giglio et al., 2018; <https://doi.org/10.1016/j.rse.2018.08.005>) accessed from AppEEARS (<https://lpdaacsvc.cr.usgs.gov/appeears/>, last access: 11 February 2022), historical data of fire danger indices from the Copernicus Emergency Management Service (Copernicus, 2019; <https://doi.org/10.24381/cds.0e89c522>) accessed from the Climate Data Store (<https://cds.climate.copernicus.eu/>, last access: 11 February 2022) and UKCP18 regional climate model projections (Met Office Hadley Centre, 2018; <https://catalogue.ceda>).

ac.uk/uuid/589211abeb844070a95d061c8cc7f604, last access: 11 February 2022). The code used to analyse these datasets to produce the figures and results presented in this article and the processed data required to reproduce the plots are available on request.

Author contributions. MP and EV developed the methods and code to analyse the historical data and model projections, produced the results and plots, and wrote a first draft of the paper. RB conceptualised the research and contributed to the writing. EP reviewed the paper and contributed to the writing.

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Exploring the impacts of climate change and mitigation policies on UK feed barley supply and implications for national and transnational food security



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Abstract

The destabilizing impacts of climate change make mitigation measures a global imperative and urgent. The Paris Agreement and the Sustainable Development Goal 13 give fresh impetus for climate change mitigation action. However, mitigation policies can have unintended consequences for food security. The current study assessed the impacts of UK climate mitigation policies on future food penalties (using feed barley and meat) and to highlight potential tele-coupled implications for transnational food security (especially in developing economies). Total barley production for the 2030s, 2040s, and 2050s was obtained from simulated barley yields under three emissions scenarios (high, medium, and low; or HES, MES, and LES, respectively) and projected land area for barley production from the updated study on UK land use change under climate mitigation policies. Future demand for feed barley was obtained as a product of projected population and per capita demand. Feed barley equivalent meat production and demand were obtained using similar approach. The differences between future demand and supply were calculated for all combinations of climate change and land use scenarios. The results show that land area under climate mitigation scenarios would be 82–87% of the mean land area currently under barley production. Consequently, deficits in feed barley supply could range from 40 to 51% of total demand from the 2030s to the 2050s, while deficits in meat could be as high as 71% of total demand in the 2050s. As a result, the UK moves from its current position of net exporter of barley to a major net importer of barley or meat in the future. If current land area is maintained, maximum feed barley and meat deficits would be 44% and 65%, respectively, of total demand in the 2050s. For example, based on the middle of the road land use and climate change scenarios, together with projected 90th percentile yield, feed barley deficits in the UK would be 7102, 7548, and 7963 thousand tonnes for the 2030s, 2040s and 2050s, respectively. The main sources of export of barley to the UK would have either excess domestic demand over supply or inadequate surplus to meet the fractional exports required to serve the UK deficit. Potentially increased imports by the UK from Ukraine, Germany, and France would require an adjustment in the trade flows to the current top importers from these countries as the UK's share of import would increase from less than 1% currently to 13%, 32%, and 183%, respectively. The observed magnitude of deficits highlights adverse impacts of UK land-based climate mitigation policies on its food security, with potential consequences for transnational food security, especially in developing economies that rely on the global market for feed barley or meat supply.

Keywords Climate change mitigation · Food security · Barley · Land use change · UK

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1 Introduction

Grains constitute the largest source of energy in animal feed. Future demand for feed use of grains is projected to increase significantly in response to projected increase in demand for meat and animal products [1]. Feed use accounts for about 35% of total grains (largely coarse grains) produced globally, and about 1.1 billion tonnes of grains will be required for animal feed by 2050 [1, 2]. In the UK and the EU, more than half of total grains produced is used for animal feed [2, 3]. Globally, barley (*Hordeum vulgare* L.) is the largest coarse grain used as animal feed, and feed use accounts for about 53% of total barley produced while the remainder goes into mainly malting [4]. By 2050, developing countries would account for about 56% of global feed use of coarse grains [1], indicating a potentially large increase in demand for or imports of feed barley. Kruse [5] reported that, by 2050, global aggregate barley production will have to increase by 54% (over 2000 level) to serve projected demand. In terms of area harvested and quantity produced, barley is the second most important crop in the UK and the number one crop in Scotland.

Apart from genetic and agronomic factors, climate change and land use would be the principal determinants of total barley production in the future. In the UK and other northern temperate environments, elevated atmospheric carbon dioxide in combination with moderate warming and adequate soil moisture would likely benefit cereal crops such as barley [6–8]. Such conditions would likely enhance photosynthetic capacity through efficient radiation capture and water use, and thereby increase biomass production, harvest index and drought tolerance [9–14]. In the UK and Ireland, for example, it has been shown that barley production would remain viable under climate change and yield would increase with higher carbon emission scenarios [13, 15]. Hence, the UK could be considered a climate change beneficiary country in terms of barley production. Land use, however, could constrain total production depending on the trajectory and impacts of climate change mitigation policies on area of croplands [16].

Impact assessment of climate change on food security and adaptive responses have been focused largely on local to national scales. However, due to international trade, agricultural surpluses and deficits are increasingly becoming tele-coupled, capable of causing environmental and economic impacts not only in the countries directly involved in the trade but also across other geographical and temporal scales. Even so, risk of tele-coupled impacts of mitigation measures in a given country on other countries has yet to be factored into the discourse on climate change adaptation and justice, especially in developing countries. Food security in developing countries will be

impacted by not only agro-climatic conditions but also the socio-economic and political circumstances that permit access to food on the global market [17, 18]. This implies the need to assess and monitor the future production of crops such as barley in climate-beneficiary countries and the potential impacts on especially developing countries that depend on such crops for food security. The impacts of such future production levels on supply and prices on the global market are an important signal for food security adaptation in distal, dependent (developing) countries.

There have been reports on the potential adverse effects of climate mitigation efforts on food security (e.g. [19–22]). Climate change mitigation affects food security in two principal ways: (1) competition for resources (land, water, inputs) between food crops and mitigation plants (energy crops, forestry), and (2) competition for food crops between energy production and human consumption or feed use. A third dimension is the cost of mitigation, which could force reductions in resource allocation to food production or reduce economic welfare. A probable outcome of this competition is higher food prices (due to supply deficit) or reduced real income which would affect access to food [20]. Based on a multi-model assessment of the combined effects of climate change and mitigation policies on food security, Hasegawa et al. [23] reported that, by 2050, stringent mitigation policies, implemented evenly across all sectors and regions, would have a larger negative effect on global hunger and food consumption than the sole impacts of climate change. Low-income countries in Africa and South Asia would be severely affected. A related study indicates that, by the end of the century, land-based mitigation activities could potentially increase food prices by 110% or more over baseline (2005) value, with Africa and Asia suffering the greatest price effects [24]. Richardson et al. [18] and Krishnamurthy et al. [25] reported that current geographical distribution of vulnerabilities to food insecurity would remain unchanged or worsen under climate change unless extreme combinations of mitigation and adaptation measures are applied. While attention has been drawn to the potential impact of climate mitigation on food supply and/or prices at larger spatial scales, these reports have focused mostly on the potential impacts of increased bioenergy consumption on the risk of hunger (e.g. [19, 21, 22]). Hasegawa et al. [20] included the effect of cost of mitigation on the risk of hunger. In the current study, we assess the combined effect of climate mitigation policies (through land use change), climate change, and population change on national and transnational food security using crop- and country-specific case. We use future barley demand and supply in the UK (a strong trading nation with high self-sufficiency rate with respect to barley) as a case to demonstrate quantitatively the potential adverse impact of mitigation policies on national food

security and the implications for tele-coupled impacts on the food security of distal economies, especially developing countries that could be considered as climate change victims.

2 Materials and methods

2.1 Future barley yields and production

2.1.1 Climate change

Projected climate change datasets for three emissions scenarios (low, medium and high, or LES, MES, and HES), which correspond to the B1, A1B, and A1FI emission scenarios, were obtained from the UKCP09 database. The UKCP09 is a publicly accessible online database that provides projected climate change datasets over the UK to the end of the century [26]. In the UKCP09, the future climate data have been downscaled to UK administrative regions and river basins at a grid resolution of 25 km, and averaged over seven overlapping 30-year time slices [26]. The Weather Generator embedded in the UKCP09 applies a stochastic approach to a user-specified number of model variants that are randomly sampled from the probabilistic projections to generate statistically credible future climate variables at daily or hourly time scales [27]. The Weather Generator was used to generate daily climate data for the 2030s, 2040s, and 2050s for the fourteen UK administrative regions and for each of the three emissions scenarios. Each Weather Generator request comprised 100 random samples from 10,000 model variants run for a duration of 30 years.

It is acknowledged that the UKCP09 has been superseded by the UKCP18 (UK Climate Projections 2018), which was not published at the time of the current study. Principally, the UKCP18 has five emission scenarios (RCP2.6, RCP4.5, RCP6.0, RCP8.5, and SRES A1B). While the use of the RCP emission scenarios represents an inclusion of a wider range of future atmospheric CO₂ concentrations and associated climate forcing over the SRES used in UKCP09, large overlap of projected ranges of several climate metrics exists between UKCP18 and UKCP09 [28]. For example, the range of future climate change forcings of RCP4.5 to RCP8.5 is similar to the scenarios of the UKCP09 [28]. The explicit inclusion of the SRES A1B in the UKCP18 permits comparison of UKCP18 and UKCP09 to obtain the effects of the improvement in methodology and input data. Such a comparison for South East England, for example, shows that there is not much difference in summer temperatures between the UKCP18 and UKCP09 even though the

difference is slightly larger for winter temperatures [28]. With precipitation, the UKCP18 tends to show either larger reductions or smaller increases over baseline values compared to UKCP09. Generally, the two datasets show considerable similarities over land [28]. These notwithstanding, using UKCP18 instead of UKCP09 could alter the results as reported in the current study.

Soil data were obtained from the Crop Growth Monitoring System (CGMS) database in the new Soil Information System (SINFO), which is part of the agricultural monitoring and yield forecasting programme of the European Union [29]. After download, the SINFO data were imported and processed in ArcGIS 9.1 (ESRITM, Redlands, CA, USA). The UK was clipped and attribute tables of soil mapping units (SMU) and soil typological units (STU—describing soil properties) were joined using common fields. The resulting join table was in turn joined to the soil physical group attribute table to obtain a single attribute table for the soil polygons. Weighted averages of soil hydraulic properties were then obtained at regional level. It is noteworthy that drainage characteristics such as drainage coefficient, saturated hydraulic conductivity, and curve number for surface runoff (not available from the SINFO data) were generated in the AquaCrop model based on the input values of saturated water content, field capacity, and permanent wilting point which were obtained from the SINFO data. The soil hydraulic properties input data required by AquaCrop and were obtained from the SINFO are presented in “Appendix 1”.

The crop data were based on the genotype Westminster, which was grown in the field at The James Hutton Institute (Invergowrie, Dundee, UK) and used in the initial calibration and validation of the AquaCrop model. During this study, the genotype Westminster was on the HGCA (Home-Grown Cereals Authority, UK) Recommended List as a high-yielding genotype, widely grown both as spring and winter barley and for feed and malt. The AquaCrop model which was used in the current study simulates the responses of crop biomass production and yield in response to mainly soil and climatic conditions. The detailed description of the model's architecture and algorithms can be found in Raes et al. [30] and the conceptual framework, underlying principles and distinctive components and features can be found in Steduto et al. [31]. For the calibration, the difference between observed and predicted yield was 0.14 tons ha⁻¹ while the difference at validation using 2 years data points were 0.91 and 1.73 tons ha⁻¹. The overall calibration–validation root-mean-square error (RMSE) for genotype Westminster was 8.10% (see [32] for further details).

Detailed description of the simulation under projected climate change can be found in Yawson et al. [15]. In summary, the search for future sowing dates for UK regions was based on the HGCA recommended sowing window (from late January to end of February in the south and east of England, and from late February to the end of March in Scotland). Because of potential quality and yield penalties outside of this window, the search for appropriate sowing dates in AquaCrop was restricted to the HGCA recommendation. To this end, the method of Mainuddin et al. [33] was followed to obtain sowing dates. Thus, sowing date for each region was obtained by forcing AquaCrop to the 1990 spring barley yield for that region by varying only sowing date until the simulated yield approximated the observed yield. The first sowing date that gave the closest match between observed and simulated yields was taken as the sowing date as was reported. For 1990, the differences between the observed and simulated yields ranged from -0.14 to 0.59 tons ha^{-1} . The obtained sowing dates were applied to simulate and compare the yields from 1980 to 1989 in order to assess the reliability of the sowing dates using the RMSE [15, 33]. The RMSE values for the 14 UK regions ranged from 0.44 (for East of England) to 1.15 tons ha^{-1} (for Wales), and 0.35 tons ha^{-1} for the UK [15]. Subsequently, the projected climate data, together with the soil, crop, and atmospheric CO_2 concentration data for the respective emission scenarios and time slices were used to simulate future spring barley yields using the sowing dates obtained for the respective regions. The simulations were done under rainfed conditions, based on thermal time, and assuming no fertility stress. The main AquaCrop parameterization information is provided in “Appendix 2”. The regional yields and their associated statistics are reported in Yawson et al. [15]. The regional yields were averaged to obtain the UK national yield data used in the current study. Further information can be obtained in Yawson [32].

2.1.2 Climate change mitigation and land use

Future cropland data were obtained from the study report *Inventory and Projections of UK Emissions by Sources and Removals by Sinks due to Land Use, Land Use Change and Forestry (LULUCF)* by Thomson et al. [34]. This is a revised or updated study on land use responses to climate mitigation policies and aspirations in the UK. The UK Climate Change Act [35] targets 80% emission reduction (relative to the 1990 baseline) by 2050. For purposes of carbon budgets under the UK Climate Change Act [35], European Union Monitoring Mechanism and the UN Framework Convention on Climate Change (UNFCCC), it is mandatory for the UK to report projections for the Land Use, Land Use Change and Forestry (LULUCF) sector [36]. The UK Climate

Change Act specifically requires periodic reports on progress towards the emission target and review of strategies and policies to remain on track as new inventory data become available. It is in this context that the studies by Thomson et al. [34, 36] were conducted for the UK Government. The latest report on projected emissions and removals of greenhouse gases from the LULUCF sector [34] differs from the 2013 report in terms of scenarios, data currency, and methodological considerations [34]. In the updated report, three policy scenarios (Central, Low, and Stretch) and two baseline scenarios were developed and the projected cropland areas were different from those in the previous report. It is important to note that these policy scenarios were jointly developed by the Department of Business, Energy and Industrial Strategy (BEIS, formerly the Department of Energy and Climate Change), the Department for Environment, Food and Rural Affairs (Defra), and the Forestry Commission, in consultation with the UK devolved administrations and incorporate current land use policies and aspirations [34]. Details of the scenarios are available in Thomson et al. [34]. In summary, the Baseline 1 scenario (BL1) maintains the 2000–2009 average activity rates (based on climate change-related and forestry policies prevalent in July 2009) to the 2050s. The Baseline 2 (BL2) is similar to BL1 except that forest planting rates decline to a low level after 2015 as grant-aided planting remains constant at 2009 level. In the Central Scenario, policies and funding prevalent in 2014 remain unchanged to 2050 for non-forest activities. However, the 2014 afforestation rates are maintained to 2021 and reduced thereafter to the levels in BL2. The Low Scenario projects forward the climate change mitigation policy aspirations for the UK Devolved Administrations beyond 2021. Finally, the Stretch Scenario is an ambitious scenario which captures climate change mitigation actions exceeding current policy aspirations or funding. The results were provided for England, Northern Ireland, Scotland, and Wales, and national (UK) average. This updated study has implications for the quantity of food that can be produced due to land use changes.

Projected areas of UK (national) croplands for the 2030s, 2040s, and 2050s, due to land use changes in response to emission targets or climate mitigation policies are obtained from Annex 2 [34] and the corresponding areas of barley were obtained as a proportion of the croplands. To achieve this, average area of barley for the period 2000–2012 was calculated as a percentage of the mean of total croplands for the same period. This proportion (16.36%) was applied to the areas of croplands obtained from Thomson et al. [34] to obtain future areas of barley for the respective climate mitigation scenarios and time slices. In addition, the actual mean value of area under barley cultivation for the period 2000–2012 (1026 thousand

ha) was used to represent land area under current conditions or a business as usual (BAU) scenario, which remains unchanged from the 2030s to the 2050s.

Unlike the previous study [16], the current study uses all the land use scenarios from Thomson et al. [34] and does not incorporate crop-specific land use changes in response to non-policy factors. Future total barley production (supply) was obtained as the product of simulated yield [15] and projected land area of barley for the respective time slices, emissions and land use scenarios. In the current study, the 90th and 50th percentile yields were used to represent the optimistic situation (highest yields) and to make the current paper comparable to the previous paper based on the mean or median (normal) yields.

2.2 Future demand, balances and implications for national and transnational food security

Future demand of feed barley and/or feed barley equivalent meat demand (FBEM) and differences between demand and supply were estimated as described previously [16]. In summary, the projected population from the constant fertility scenario was used in combination with projected per capita feed barley or meat demand to obtain future total demand. Projected per capita demand for meat and feed grains were obtained from the Comprehensive Assessment of Water Management in Agriculture [37] as described in Yawson et al. [16]. The IWMI [37] study was based on assumptions of changes in population, dietary requirements and composition (or preferences), incomes, prices of commodities and inputs, and urbanization. It explicitly provided projected demand for feed grain and meat and was therefore more suited to the purpose of the current study unlike other similar reports such as the FAO projections [1], The Global Harvest Initiative Report [5], and Tilman et al. [38].

A causal loop analysis, a component of systems dynamics analysis, was used to explore the implications of the future barley or FBEM balances in the UK on national food security and potential tele-coupled impacts on the food security of distal economies through cascading effects on the regional or global food market. Causal loop diagram (CLD) analysis was the main tool used to conceptualize the national food security risk and then for exploring the broader implications for transnational food security risks (Fig. 4). The CLD approach helps provide nuanced understanding of the structure (system components, connections and interactions) and overall behaviour of a system or problem by making explicit the causal-feedback relationships between components and thereby helps address the problem by evaluating outcomes [39, 40]. The CLD, a graphical depiction of the causal-feedback relationships arising from the behaviour and interaction of system

components or other exogenous factors relevant to the system, is the first major stage of constructing a system dynamics model.

To provide numerical support or a perspective to what the CLD portrays, an analysis of future barley supplies from the main trading partners of the UK, relative to the UK deficits under the medium emissions climate change scenario (MES) and the Central climate mitigation land use scenario in the 2050s was done. From the FAOSTAT database, eight countries accounted for 95% of total UK barley import. These countries were therefore selected to represent the potential sources of UK barley imports in the future. Taking these eight countries as 100%, their respective percentage shares in total UK barley imports were calculated: Ireland (47%), France (17%), Germany (13%), Ukraine (7%), Spain (5%), Denmark (4%), Sweden (4%), and Italy (3%). Subsequently, the national yields of these countries, together with UK yields, for the period 1998–2017 were retrieved from the FAOSTAT database. These were used to develop regression relationships to predict future yields in the trading partner countries but this proved unreliable. As a result, the mean of the yearly difference between each country's yield and the UK yield for the period under consideration (M) was calculated and added to 90th or 50th percentile UK yield under the MES to obtain an estimated yield for the partner country in the 2050s:

$$M = \pm \sum_{i=1}^n \frac{YP_i - YU_i}{n} + YU_{\text{MES},2050}$$

where M = mean of the differences in yields for the period under consideration; n is the number of observations or years; YP_i is the yield in partner country i for year i ; YU_i is the UK yield for year i . $YU_{\text{MES},2050}$ is the UK yield under the medium emission scenario in the 2050s.

The estimated yields were then multiplied with the mean land area of barley for the same period to estimate the future production (i.e. assuming the partner countries would maintain their current land area to the 2050s). From the food balance sheets of the partner countries for 2016, the per capita supply of barley was obtained from the food balance sheet as the ratio of domestic supply to population. This per capita supply was then used as the status quo demand and multiplied by the projected population of the respective partner country [41] to obtain future demand. The estimated future demand was then subtracted from the future production to obtain the 'exportable fraction' (surplus over supply for domestic use or demand). Because Ireland, France, Germany and Ukraine accounted for 84% of total UK barley import, the subsequent analysis was based on these countries. For these countries, the difference between the future exportable fraction and current total export was calculated. The main export destinations

of barley from these four countries were retrieved from the FAOSTAT for the year 2016. The percentage share of the top 20 importing countries from the total export of these four countries was calculated. The resulting value was used to calculate the share of export to the top 20 countries from the future exportable fraction. The remaining quantity of barley (residual) for export to the rest of the world (including the UK where applicable) was then obtained as the difference between the exportable fraction and the quantity to be exported to the top 20 importing countries.

3 Results

3.1 Future barley production

Projected barley yields were highest under the high emissions scenario (HES) and in the 2050s and lowest under the low emissions scenario (LES) in the 2030s (Fig. 1).

Generally, both the 90th and 50th percentile yields increased across the time slices. The highest 90th and 50th percentile yields were 8.27 and 7.87 tonnes ha⁻¹, respectively, under the HES in the 2050s. The lowest yield value was 6.01 tonnes ha⁻¹ (the 50th percentile yield under the LES) in the 2030s.

The total area of land for barley under the business as usual (BAU) or current condition was 1026 thousand ha. For all climate mitigation scenarios and time slices, the projected land area was lower than the current land area under barley production (the BAU). The largest land area (approximately 893 thousand ha under the stretch scenario in the 2050s) was 87% of the land area under the BAU, while the lowest land area (approximately 840 thousand ha under the BL scenarios in the 2030s) was about 82% of the BAU (Fig. 2). The projected land areas under the low and central scenarios were similar, and the value in the 2050s (884 thousand ha) was 86% of the BAU. Because

Fig. 1 Projected barley grain yield for the low, medium, high emission scenarios (LES, MES, and HES, respectively) in the 2030s, 2040s, and the 2050s

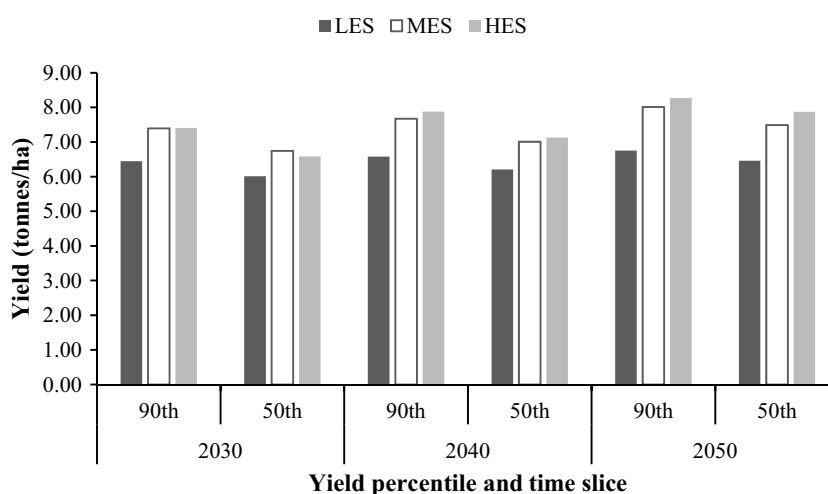


Fig. 2 Projected land area for barley in 2030, 2040, and 2050 based on climate mitigation policies [34]. The total area of barley under the BAU remains constant at 1026 thousand ha from 2030 to 2050

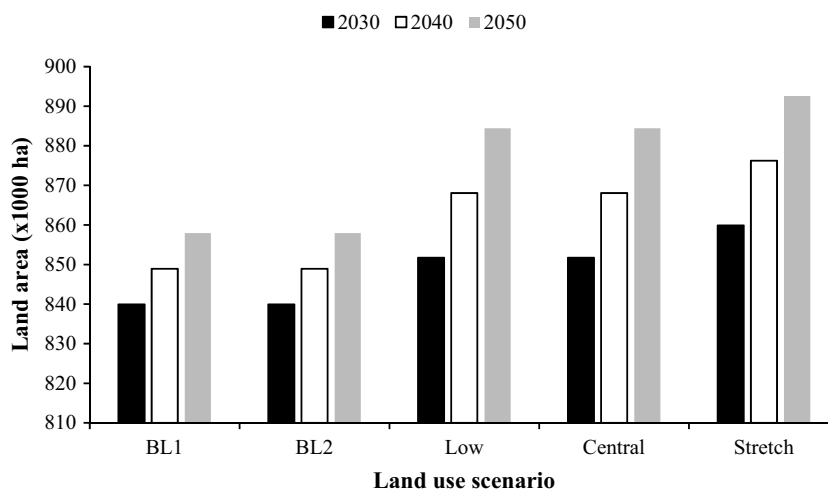


Table 1 Total barley production (thousand tonnes) under projected climate and land use change

Emission scenario	Yield percentile	BAU	BL	Low	Central	Stretch
2030						
LES	90th	6614.27	5414.69	5490.63	5490.63	5543.36
	50th	6167.93	5049.30	5120.11	5120.11	5169.28
MES	90th	7584.74	6209.15	6296.23	6296.23	6356.70
	50th	6917.97	5663.31	5742.73	5742.73	5797.89
HES	90th	7602.66	6223.83	6311.11	6311.11	6371.72
	50th	6752.27	5527.66	5605.18	5605.18	5659.02
2040						
LES	90th	6748.52	5583.77	5709.68	5709.68	5763.48
	50th	6369.71	5270.35	5389.18	5389.18	5439.97
MES	90th	7874.1	6515.09	6661.99	6661.99	6724.77
	50th	7188.56	5947.87	6081.98	6081.98	6139.29
HES	90th	8082.00	6687.11	6837.89	6837.89	6902.33
	50th	7315.12	6052.59	6189.06	6189.06	6247.38
2050						
LES	90th	6928.15	5793.16	5972.13	5972.13	6027.36
	50th	6627.28	5541.58	5712.77	5712.77	5765.61
MES	90th	8215.91	6869.97	7082.20	7082.20	7147.70
	50th	7685.72	6426.63	6625.16	6625.16	6686.44
HES	90th	8486.22	7095.99	7315.20	7315.20	7382.86
	50th	8079.69	6756.06	6964.77	6964.77	7029.18

the cropland areas for the BL1 and BL2 were the same, we simply use one to represent the baseline (BL) henceforth.

Future total barley production for all climate and land use change scenarios across the time slices are presented in Table 1. At the 90th percentile yield, total barley production under the land-based climate mitigation policies ranged from 5415 (LES, 2030s) to 7383 thousand tonnes (HES, 2050s), which were approximately 82–87% of the production under the BAU. For the 50th percentile yield, total barley production ranged from 5049 (LES, 2030s) to 7029 thousand tonnes (HES, 2050s). For these scenarios, highest production occurred under the Stretch scenario while the lowest occurred under the baseline (BL) scenario in correspondence with differences in land area. As yields were highest in the 2050s, total production was highest in the 2050s. For the BAU and BL, the total production under the LES did not differ substantially between the 2030s and the 2040s but that is not the case for the other land use and emissions scenarios.

Total amount of feed barley that would be available from total domestic barley production (assuming current proportionate distribution to end uses remains unchanged) is presented in Table 2. Since this is a constant proportion of the total production values, the pattern is similar to the total production. That is, total amounts of feed barley from domestic production under the climate mitigation land use scenarios are lower than the

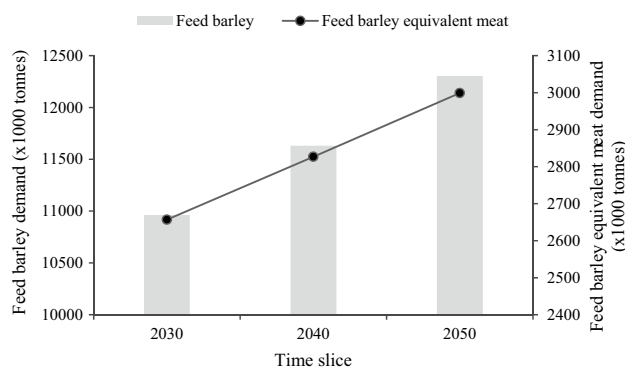
corresponding values under the BAU. If current land area under barley cultivation (BAU) is continued, then total feed barley supplies from domestic production at 90th percentile yield would range from 4055 (LES, 2030s) to 5202 thousand tonnes (HES, 2050s), which would be 18–13% higher than the corresponding supplies under the climate mitigation land use scenarios.

3.2 Future barley demand and deficits

Projected UK population for the 2030s, 2040s, 2050s were 71.9, 76.1, and 80.3 million. The corresponding demand ranged from 10,962, to 12,304 thousand tonnes for feed barley, and 2657–2999 thousand tonnes for feed barley equivalent meat (Fig. 3). At the 90th percentile yield, the least and largest deficits in feed barley under the current land area (the BAU) ranged from 6302 (HES) in the 2030s to 8057 thousand tonnes (LES) in the 2050s (Table 3). These represented about 31% and 40% deficits, respectively, in total demand in the 2030s or 31% and 44%, respectively, of total demand in the 2050s. Under the climate mitigation land use scenarios, the least and largest deficits were 42% (Stretch, HES) and 51% (BL, LES), respectively, of total demand in the 2030s. The corresponding values in the 2050s were 40% and 53%, respectively. Using the Central scenario of the climate mitigation policies, the middle of the road emissions scenario (MES), and the 90th percentile

Table 2 Proportionate feed barley supply (thousand tonnes) from domestic production

Emission scenario	Yield percentile	BAU	BL	Low	Central	Stretch
2030						
LES	90th	4054.55	3319.21	3365.76	3365.76	3398.08
	50th	3780.94	3095.22	3138.63	3138.63	3168.77
MES	90th	4649.44	3806.21	3859.59	3859.59	3896.66
	50th	4240.71	3471.61	3520.30	3520.30	3554.12
HES	90th	4660.43	3815.21	3868.71	3868.71	3905.87
	50th	4139.14	3388.46	3435.98	3435.98	3468.98
2040						
LES	90th	4136.84	3422.85	3500.03	3500.03	3533.01
	50th	3904.63	3230.73	3303.57	3303.57	3334.70
MES	90th	4826.82	3993.75	4083.80	4083.80	4122.28
	50th	4406.59	3646.05	3728.26	3728.26	3763.39
HES	90th	4954.27	4099.20	4191.63	4191.63	4231.13
	50th	4484.17	3710.24	3793.89	3793.89	3829.64
2050						
LES	90th	4246.96	3551.21	3660.92	3660.92	3694.77
	50th	4062.52	3396.99	3501.93	3501.93	3534.32
MES	90th	5036.36	4211.29	4341.39	4341.39	4381.54
	50th	4711.35	3939.52	4061.23	4061.23	4098.79
HES	90th	5202.05	4349.84	4484.22	4484.22	4525.69
	50th	4952.85	4141.46	4269.40	4269.40	4308.89

**Fig. 3** Projected feed barley and feed barley equivalent meat demand

yield, the projected deficits in feed barley supply would be 7102, 7548, and 7963 thousand tonnes for the 2030s, 2040s and 2050s, respectively. The deficits could far exceed 8000 thousand tonnes for the 50th percentile yield and the climate mitigation land use scenarios under the LES in the 2040s and 2050s and could be as high as 8907 thousand tonnes. The deficits were highest under the BL for all emissions scenarios and time slices, and at both the 90th and 50th percentile yields.

The projected deficits in feed barley equivalent meat (FBEM) demand are expectedly larger under the climate

mitigation land use scenarios than under the current land area (the BAU). At the 90th percentile yield, the FBEM deficits would range from 1527 thousand tonnes (HES, 2030s) to 1964 thousand tonnes (LES, 2050s) for the BAU (Table 4), representing 57% and 65%, respectively, of total demand in these two time slices. With the climate mitigation scenarios, the FBEM deficits ranged from 1710 to 2133 thousand tonnes from the 2030s to the 2050s. The range of the deficits represented 64% and 71%, respectively, of total demand from the 2030s to the 2050s. Using the Central scenario, the MES and the 90th percentile yield, the FBEM deficits would range from 1722 (2030s) to 1941 thousand tonnes in the 2050s. Again, these deficits increase substantially under the 50th percentile yield in accordance with the difference between the 90th and 50th percentile yields.

3.3 Implications for national and transnational food security

Causal loop diagrams (CLD) are graphical representations used to logically explore the causal-feedback relationships between components of complex systems. It is extensively used in systems dynamics analysis and modelling. This simplified CLD (Fig. 4) shows the most relevant feedback relationships between UK barley production, UK demand and supply of feed barley or FBEM, and the impacts on

Table 3 Projected deficits in feed barley (thousand tonnes) based on domestic feed barley supply

Emission scenario	Yield percentile	BAU	BL	Low	Central	Stretch
2030						
LES	90th	6907.45	7642.79	7596.25	7596.25	7563.92
	50th	7181.06	7866.78	7823.37	7823.37	7793.23
MES	90th	6312.56	7155.79	7102.41	7102.41	7065.34
	50th	6721.29	7490.39	7441.7	7441.7	7407.89
HES	90th	6301.57	7146.80	7093.29	7093.29	7056.13
	50th	6822.86	7573.54	7526.02	7526.02	7493.02
2040						
LES	90th	7495.16	8209.15	8131.97	8131.97	8098.99
	50th	7727.37	8401.28	8328.43	8328.43	8297.30
MES	90th	6805.18	7638.25	7548.2	7548.2	7509.72
	50th	7225.41	7985.96	7903.75	7903.75	7868.61
HES	90th	6677.73	7532.80	7440.37	7440.37	7400.87
	50th	7147.83	7921.76	7838.11	7838.11	7802.36
2050						
LES	90th	8057.05	8752.79	8643.09	8643.09	8609.23
	50th	8241.48	8907.01	8802.07	8802.07	8769.68
MES	90th	7267.64	8092.71	7962.61	7962.61	7922.46
	50th	7592.65	8364.48	8242.78	8242.78	8205.21
HES	90th	7101.95	7954.16	7819.78	7819.78	7778.31
	50th	7351.15	8162.54	8034.60	8034.60	7995.11

Table 4 Projected deficits in feed barley equivalent meat (FBEM, thousand tonnes) due to deficits in domestic feed barley supply

Emission scenario	Yield percentile	BAU	BL	Low	Central	Stretch
2030						
LES	90th	1674.25	1852.48	1841.2	1841.2	1833.36
	50th	1740.57	1906.77	1896.25	1896.25	1888.94
MES	90th	1530.05	1734.44	1721.5	1721.5	1712.52
	50th	1629.12	1815.54	1803.74	1803.74	1795.55
HES	90th	1527.39	1732.26	1719.29	1719.29	1710.29
	50th	1653.74	1835.7	1824.18	1824.18	1816.18
2040						
LES	90th	1821.6	1995.12	1976.36	1976.36	1968.35
	50th	1878.03	2041.82	2024.11	2024.11	2016.55
MES	90th	1653.91	1856.37	1834.49	1834.49	1825.13
	50th	1756.04	1940.88	1920.9	1920.9	1912.36
HES	90th	1622.93	1830.74	1808.28	1808.28	1798.68
	50th	1737.18	1925.28	1904.95	1904.95	1896.26
2050						
LES	90th	1963.84	2133.42	2106.68	2106.68	2098.43
	50th	2008.79	2171.01	2145.43	2145.43	2137.54
MES	90th	1771.43	1972.53	1940.82	1940.82	1931.04
	50th	1850.65	2038.77	2009.11	2009.11	1999.95
HES	90th	1731.04	1938.76	1906.01	1906.01	1895.9
	50th	1791.78	1989.55	1958.37	1958.37	1948.74

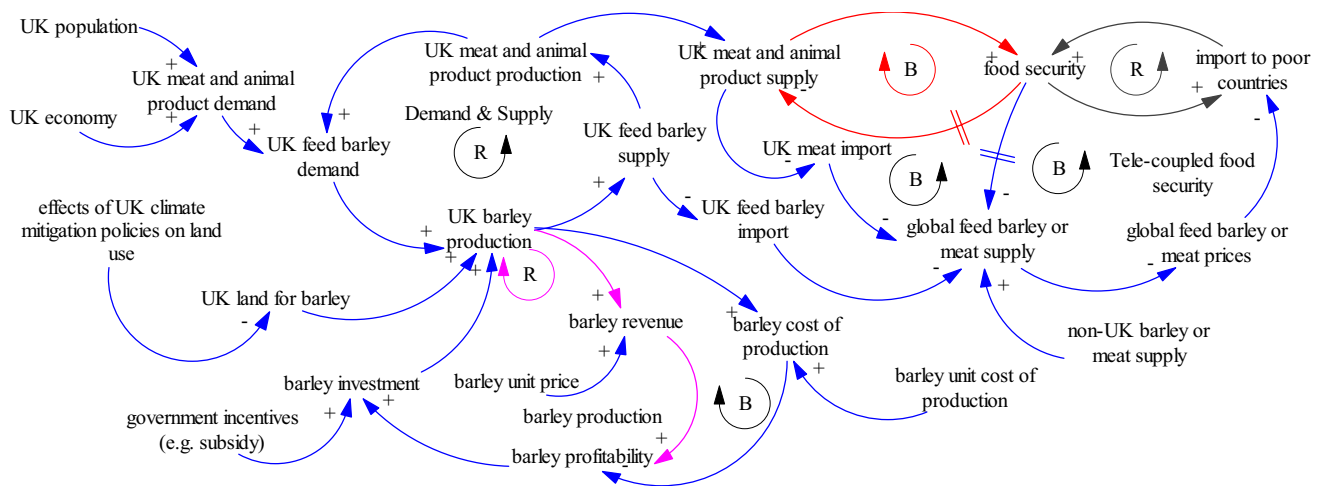


Fig. 4 Causal loop diagram of tele-coupled impacts of UK feed barley deficits on the food security of distal, poor economies

national and transnational food security. In the CLD, a negative sign indicates opposite direction (i.e. an increase in one variable results in a corresponding decrease in the variable to which it is linked and vice versa). Conversely, a positive sign indicates change in the same direction (i.e. a decrease or increase in one variable results in a corresponding decrease or increase, respectively, in the variable to which it is linked). In the CLD, there are three reinforcing feedback loops (indicated with R) and four balancing loops (indicated with B).

In the demand and supply loop or sub-system (upper left), increase in population and economic welfare (exogenous factors) leads to increase in demand for meat and animal products, which triggers increased demand for feed barley. In response, barley production increases to match the demand, which leads to increased feed barley supply and therefore meat production. Increased barley production leads to increased barley revenue, profitability and investment in barley production (second reinforcing loop in the barley production sub-system, lower left). These are positive self-reinforcing loops. However, barley production is not only driven by demand and profitability, but also subject to land area allocated to barley production in response to climate mitigation policies and the unit cost of barley production (exogenous factors which can constrain total barley production). This creates a self-balancing loop (lower left) where growth or expansion is moderated and checked from within the loop, resulting in an innately stable sub-system. This loop provides a check or limit on total barley production regardless of the magnitude of demand. Thus, the barley production sub-system is controlled by two main loops: (1) a positive self-reinforcing loop based on incentives, profitability and investments, and (2) a balancing loop driven by land area and cost of production. On the righthand side of the CLD,

feed barley or meat supply (availability) on the global market depends on the magnitude of UK feed barley or meat imports (which in turn depend on the scale of deficits in domestic supply from domestic production) and non-UK supply. Increased UK imports lowers availability on the global market, which results in increased prices of feed barley or meat, then to reduced imports to especially poor countries, with the ultimate result of reduced food security. Here, the feedback relationships between food security and global supply or availability of feed barley or meat is controlled by two balancing loops, one related to the magnitude of UK imports, and the other related to global prices and transnational imports to especially poor economies (tele-coupled impacts loop). Food security itself comprises the UK national loop (balancing loop with a delayed feedback effect on UK national feed barley and meat supply), and the transnational loop (positive self-reinforcing loop, respectively).

The analysis of the UK's barley imports showed that the UK is currently a small importer. From the top four sources of import, the UK accounted for 29,593 tonnes (approximately 99.96%) of export from Ireland, with Sweden and South Africa accounting for only 7 and 5 tonnes, respectively (approximately 0.02% each, Fig. 5). The UK ranked 37th importer, accounting for 0.01% of export from France, 19th (or 0.01%) of export from Germany, and 28th (or 0.07%) for Ukraine. The top 20 importers from France, Germany, and Ukraine accounted for 98.5%, 99.9%, and 98.4%, respectively, while the top 20 importers from the UK accounted for approximately 100% of the UK export (data not shown). About 50% of the top 20 UK importers were European countries and about 25% were North Africa Middle East (MENA) countries (Fig. 6). In 2016, the total export by the UK was 1,757,080 tonnes while the total import was 107,290 tonnes (approximately 6% of export),

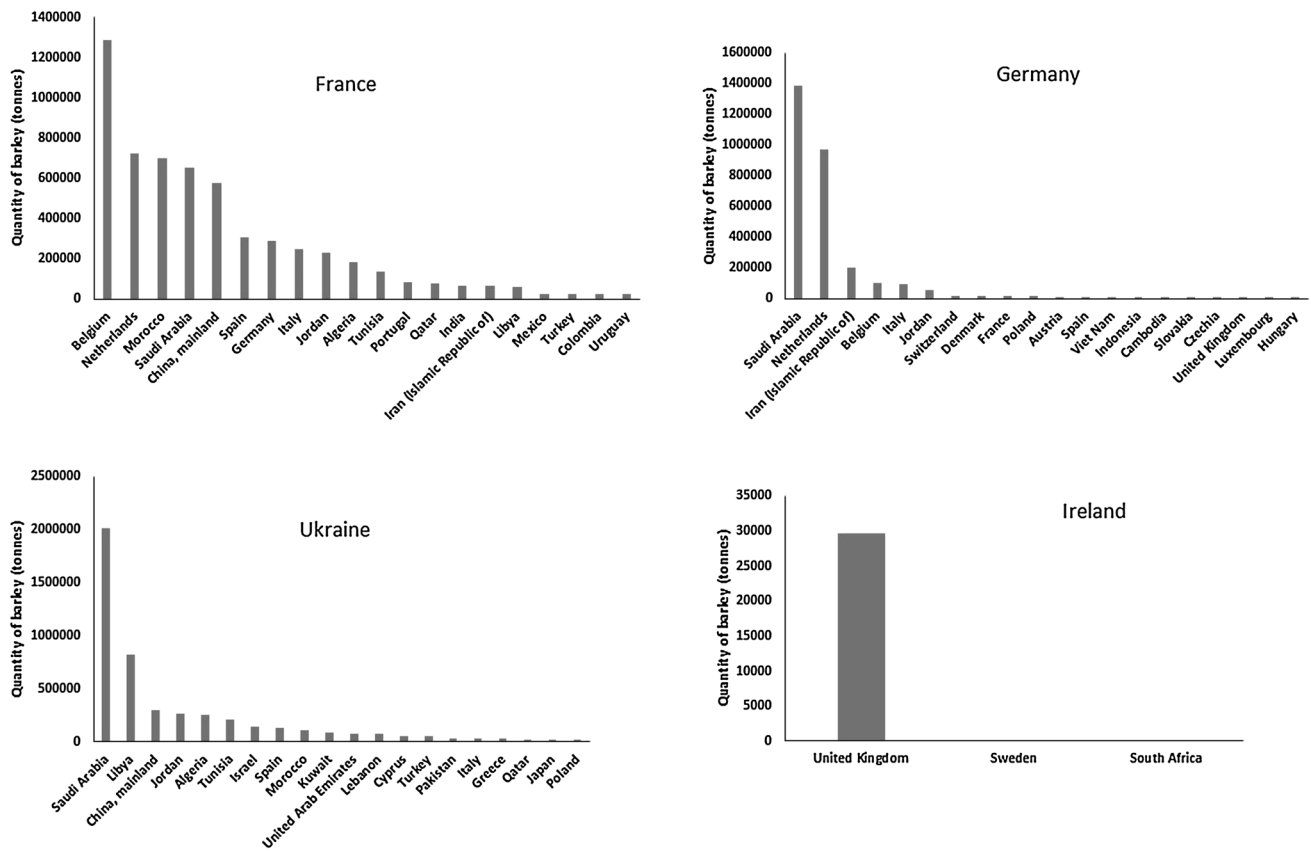
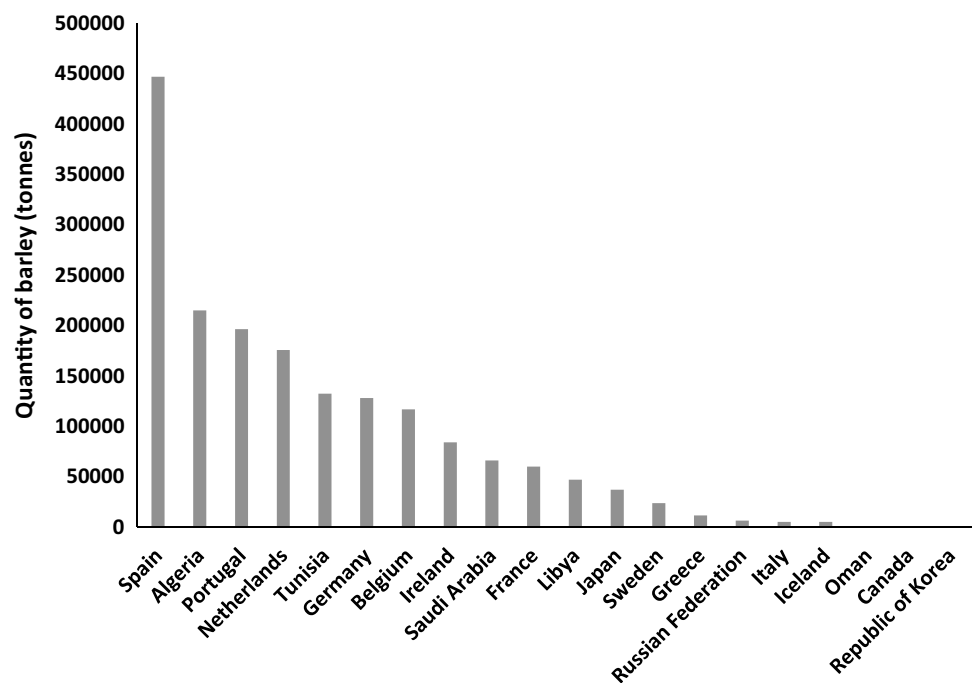


Fig. 5 Current top 20 importers of barley from the main sources of barley imports to the UK

Fig. 6 Current top 20 importers of barley from the UK



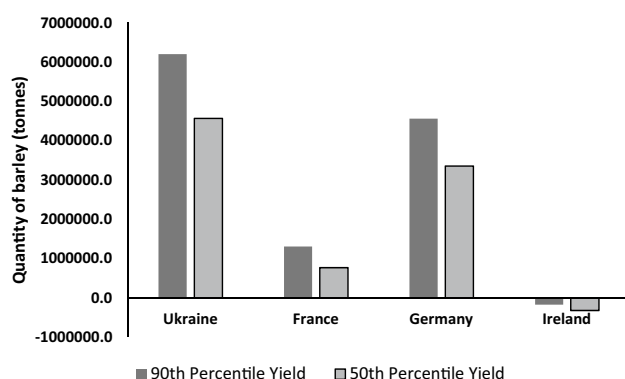


Fig. 7 Exportable fraction of barley (surplus over domestic demand that can potentially be exported) in the UK's main sources of barley imports in the 2050s

with Ireland remaining the top exporting country to the UK.

Ukraine and Germany had the largest exportable fraction after accounting for domestic demand (Fig. 7). At 90th percentile yield, the exportable fraction for Ukraine and Germany were about 6.2 and 4.6 million tonnes, respectively. Ireland (UK's main source of barley) had negative exportable fraction, indicating excess demand over supply (as its current supply for domestic use was 179% of total production, implying considerable import and re-export). The future exportable fraction of Ireland and France, at both 90th and 50th percentile yields, were lower than the current total export but the reverse was true for Germany (Table 5). Ukraine's future exportable fraction was larger than current export at 90th percentile yield but not at 50th percentile yield. After accounting for the share of export to the top 20 importers from the exportable fraction, the remaining quantity (residual) was generally insufficient to meet the expected share of export to satisfy the UK deficit. It is noteworthy that the estimates of future production and demand were based on current land area and per capita supply, respectively.

4 Discussion

4.1 Mitigation policies, land use change and UK deficits

Climate change presents enormous existential and development burdens to humanity. Despite some scepticism, effort to tackle climate change has gained traction recently. The Paris Agreement and the Sustainable Development Goal 13 have given fresh impetus to taking extraordinary measures in the interest of climate change mitigation and adaptation. However, land-based climate mitigation efforts could have adverse, unintended consequences for food security through changes in land use. The current study assessed the effect of the updated study on UK land-based climate mitigation scenarios, in combination with projected climate and population change on the balance of demand and supply of feed barley or feed barley equivalent meat (FBEM) in the UK in the 2030s, 2040s and the 2050s. Generally, climate change alone would benefit UK barley yields as the projected yields increased from the low emissions scenario (LES) to the high emissions scenario (HES) and from the 2030s to the 2050s (Fig. 1) and over baseline yields for all emissions scenarios and time slices, indicating viability of UK barley production in the future [15]. This implies a HES gives the UK the greatest yield gain across the time slices, making the UK a climate-beneficiary country with respect to barley production.

While yield increases would be a major determinant of future supply of food from crops [3], total land area allocated to agriculture would be equally an important determinant of adequate supply of food. Notwithstanding the potential beneficial effect of climate change, the results in the current study show that total UK barley production (or supply) would substantially lag behind demand if projected reductions in land area for agriculture arising from land-based climate mitigation policies or efforts materialize. For all the climate mitigation land use scenarios and time slices, the total land area for barley production is less

Table 5 Estimated future barley supply and balances in the main countries from which the UK imports barley

Partner country	(Future – current) total export (tonnes)		Share of export to top 20 importers (tonnes)		Residual for export to other countries (tonnes)		Expected export as proportion of UK deficit ('000 tonnes)	
	90th	50th	90th	50th	90th	50th	90th	50th
Ireland	–203,798	–352,648	–	–	–	–	3742	3874
France	–4,561,573	–5,100,943	1,287,101	755,822	19,601	11,510	1354	1401
Germany	1,661,237	457,397	4,556,028	3,353,031	3191	2349	1035	1072
Ukraine	1,403,106	–235,476	6,106,764	4,494,072	98,036	72,146	557	577

than the average land area currently under production (the BAU, Fig. 2) and the land area under the Mid-scenario (equivalent to the Central Scenario in the current study) in the 2030s in the previous report of Thomson et al. [36]. As a result, even at 90th percentile yields, deficits in both feed barley and FBEM in the current study are larger than those reported previously [16]. In the current study, the observed increase in cropland area across the time slices for all the land use scenarios (Fig. 2) implies more stringent mitigation efforts in the short term, with gradual relaxation in the medium to long term. However, the increase in land area due to the progressive relaxation of the mitigation efforts, in combination with the observed yield increase, is not sufficient to significantly reduce the deficits even in the 2050s where the highest yields occur under the HES. In addition, the comparatively lower land areas under the BL and the Low land use scenarios imply much larger deficits in the future if either current or stringent emission reduction pathways are followed. Because the revised land use scenarios embody current climate policy aspirations and activities, as well as updated greenhouse gas inventory data, the projected deficits in the current study show potential food penalties that can be incurred as a result of land-based mitigation policies in the UK. The plausibility of future reductions in cropland area in the UK derives from the fact that, historically, agricultural land use change in the UK has been driven primarily by government policy intervention, with other factors such as farm incomes, prices and land values playing secondary roles [2, 42]. It is likely, therefore, that there will be reductions in the area of UK croplands (and possibly for barley) in future due to policies regarding climate change and energy. For example, Rowe et al. [43] reported that, to meet projected energy targets, between 2.7 and 7.0 Mha of land would be required for bioenergy production in the UK by 2050.

Feed barley is the largest source of coarse grain used as animal feed in the UK and the world, and the straw, which has a higher feed value than that of wheat, is also used as a source of roughage in animal feed rations. Currently, the UK has high self-sufficiency rate in barley production, with export exceeding import (Fig. 6; [16]). Total import from the baseline Food Balance Sheet (FBS) was 115 thousand tonnes and total import in 2016 was about 6% of total export. According to FAOSTAT, the UK was the 7th largest producer and the 6th largest exporter of barley in the world in 2017. Based on the Central land use scenario of the climate mitigation policies, and at a 90th percentile yield under the medium emission scenario (MES), the UK moves from a net exporter to a potentially net importer of barley of approximately 7.1, 7.5, and 8.0 million tonnes for the 2030s, 2040s and 2050s, respectively, all being larger than total production in 2018 (6.5 million tonnes). These figures are even higher under the BL climate mitigation

land use scenario. To bridge this deficit, barley area needs to increase by over one million ha by 2050 if yields do not increase substantially above those observed in this study. Meat is a major source of calories in the UK diet. According to the baseline FBS, meat consumption per capita in the UK was 84.2 kg year⁻¹, delivering 457 kcal cap⁻¹ day⁻¹. It is also a major source of not only protein but also essential micronutrients. As a result, meat production is crucial to food security in the UK, at least from the perspective of supplying the needed calories and nutrition, as well as food safety. The UK is a major producer of poultry meat and the largest producer of sheep and goat in the EU at the time of this study. Although the UK is currently a net importer of meat, it is clear that domestic meat production is crucial to its food security. Domestic meat production relies heavily on domestic feed barley supply as the production of other coarse grains, such as corn, is limited. The result in the current study show that pursuit of the land-based climate mitigation policy scenarios can result in substantial feed barley deficits, which, in turn, results in correspondingly large deficits in FBEM. Under the Central land use scenario, deficits in FBEM ranged from 1719.29 thousand tonnes (HES, 2030) to 2106.68 thousand tonnes (LES, 2050s). The deficits in the current study are observed against the backdrop of very marginal increase in per capita meat demand across the time slices as shown in the methods section, suggesting population growth will be the main driver of demand.

In spite of increasing health concerns, the consumption of meat and animal products are projected to rise to the 2050s not only due to population growth but also the rich supply of high value proteins and essential micronutrients [2]. In high-income countries such as the UK, the increase in demand will mostly be driven by population growth while higher disposable incomes would drive meat consumption in low- and middle-income countries [1]. It is acknowledged that feed requirements and use efficiencies differ among animals. Hence, differences in the proportions of different meat demand might alter the total quantity of feed barley demand and, for that matter the result of the current study. For example, it has been observed that the per capita consumption of beef is reducing while poultry and pig meats are fast becoming the preferred meat in the UK (Defra, 2011) and the EU [44], and poultry has higher feed conversion efficiency than beef cattle. However, the shift from beef and lamb to poultry and pig is attributed to the relatively higher cost per unit mass of beef or lamb and the shift towards poultry and pig might change with improvements in the economy [44]. Increasing cost of feed (mainly cereal grains) has been a major driver of increasing cost of animal production and, by extension, prices of meat in the UK (Defra, 2011). Considering that, currently, the EU and the UK use 54% and

52%, respectively, of total grains produced for feed [2, 3], the continuous and increased domestic production of feed barley is critical to UK meat production and food security (local availability and economic access to consumers). In addition to the large deficits observed in the current study, projections suggest that meat and animal feed import to the EU will increase substantially in the future [3, 44]. The EU is currently the main trading partner of the UK in terms of meat and feed grains. Faced with large deficits, and all things being equal, the UK would import barley or FBEM to balance the deficits. Given the projected situation in the EU and with Brexit on the horizon, it is important to contemplate where all this import of barley or FBEM will come from to balance the observed deficits.

4.2 Implications for national and transnational food security

Food security can be defined as the “risk of adequate food not being available” [45]. Food here refers to meat (undergirded by feed barley). Proceeding from this definition, and in the light of the discussion above, the projected deficits in feed barley and FBEM in the current study, as a result of reductions in croplands in response to land-based climate mitigation policies, raise risks of future food insecurity in the UK with transnational consequences. Meat consumption delivers a substantial fraction of total calories and essential micronutrients in the UK and the world. Shifts from meat consumption to vegetarian diets are being promoted for health and environmental reasons (e.g. [46]). While this shift can reduce meat demand, meat consumption would remain an important part of diets in the near term. Here, we used CLD and analysis of the current barley imports by the UK and estimates of future supplies in its main sources of imports as a microcase to explore the implications of the observed deficits in the UK for national and transnational food security.

According to the FAOSTAT, the UK remains one of the top 10 barley exporters in the world. The analysis of the UK imports confirmed that the UK is a small importer of barley and mainly from Ireland (Fig. 5). However, the UK imports could increase considerably in the future as deficits under all climate mitigation scenarios could be as high as 53% of total demand in the 2050s while FBEM could exceed 70% of total demand. At the same time, the exportable fraction of some of its main partners would be lower than current total export at both 90th and 50th percentile yields (Table 5). Ireland (the main source of UK's import) could have negative balance for export after serving domestic demand. As a result, the expected future imports by the UK would have to come from France, Germany and Ukraine where adjustments in the trade flows to their top importing countries would be necessary to accommodate

the increased imports from the UK. Otherwise, the residual exportable fraction would not be adequate to meet their respective proportionate exports expected to address the observed deficits in the UK. At 90th percentile yield, the expected proportionate imports from Ukraine, Germany and France to satisfy the observed deficit under the MES in the 2050s (Table 5) would be approximately 9%, 23%, 104%, respectively, of the exportable fractions. The corresponding values at 50th percentile yield would be approximately 13%, 32%, 183%, respectively. This suggests considerable pressure on the proportionate distribution of exports from these countries.

The top 20 importers of these countries include a number of developing countries in North Africa and Middle East where climate change would have strong adverse effects on agriculture. According to FAOSTAT, developing countries currently among the top ten barley importers in the world are Saudi Arabia, China, Iran, Morocco, and Jordan. Major destinations of UK barley exports include Algeria, Tunisia and Saudi Arabia. These countries feature strongly on the top importing country lists for France, Germany, and Ukraine. In the Islamic countries, feed barley could be crucial for animal production as it is perhaps difficult to import frozen or processed meat from many countries due to religious considerations (e.g. halal and slaughtering of live animals for special ceremonies).

Currently, Africa and Asia have the lowest self-sufficiency rates, and Africa accounts for about 50% of direct consumption of barley as food [4]. Previous studies have highlighted the potential adverse impacts of climate mitigation policies on global or regional food security (e.g. [18, 19, 21, 23, 25]). Africa and Asia were observed to suffer the greatest effects of food insecurity due to global stringent climate mitigation policies [20, 23] or climate impacts on vulnerability to food insecurity [25] or poor combination of mitigation and adaptation [18]. Due to the UK's strong position as a producer and exporter in the world, together with the EU, it is obvious that any reduction in production and increase in demand or import would likely intensify competition on the global market. The question is, can these developing countries compete effectively with the developed, strong trading nations, such as the UK in a tight global market of barley? The results in the current study, therefore, provides a basis for a nuanced understanding of the impacts of land-based climate mitigation policies or efforts on the food security of the UK and potential telecoupled impacts on other distal economies. Meaning, the UK (as a climate-beneficiary country with regard to barley) should make efforts to sustain higher capacity for barley production to serve the climate justice principle of ‘equitable sharing of benefits and burdens’.

4.3 Assumptions and limitations

This paper is based on several assumptions and semi-empirical relationships between current and future production and feed use of barley that have implications for the results. For example, feed use as a proportion of total barley production and in meat production is neither linear nor constant. In other words, it is not deterministic. This key assumption is a limitation to the study but is one of the very few possibilities for doing such work. Policy, economic, social, and biophysical factors could alter the proportion of barley used for feed, or the composition of barley in total feed. More so, the total UK barley production is a sum of the product of winter and spring barley yields and their respective land areas. Winter barley has a higher yield than spring barley; but spring barley covers a larger area than winter barley. The current study is based on spring barley, assuming that the relatively larger area under spring barley production would compensate for the relatively higher yield of winter barley. In addition, future yields could be increased through technology (e.g. improvements in genotypes and agronomic practices) but this was not considered in the current study. While technology-driven increase in yields is feasible, it is difficult to predict. Brisson et al. [47] observed that cereal yields seem to have plateaued in Europe and sought to explain why this is the case for wheat yields. By extension, it is possible that barley yields could be substantially increased through technology if some of the identified constraints are addressed. For example, in the current study, it was observed that if future barley yields reach 10 tons ha⁻¹ or higher, together with the current land area (BAU), the observed deficits can be reduced significantly or neutralized with a little increase over the current land area (BAU). This magnitude of yield has been observed for the genotype Westminster under experimental conditions at the James Hutton Institute (Dundee, UK). Since the mid-1980s, declines in areas of croplands in high-income countries have been more than compensated for by productivity gains per unit area and very intensive use of land [1, 48]. The question is whether this trend can continue to the 2050s given the observed plateau in cereal yields. Similarly, there is considerable scope for improving feed conversion efficiencies of animals (e.g. carcass yield per unit feed) and effecting changes in the composition of meat consumption. These could alter the scale of deficits in the current study. However, as with any projection study, the key message from the current study shows a future possibility within the bounds of the assumptions and limitations of the study. Finally, estimates of future demand and supply in the UK's main trading partner countries were based on current consumption and linear relationship between their yields and the UK yields, respectively. The

scale of demand and exportable fraction in these countries could be altered by a more detailed study of the combined impacts of climate, population, and land use changes in these countries.

5 Conclusions

Climate change mitigation is a global desideratum. However, land-based climate change mitigation policies or efforts should be carefully designed and analysed to minimize unintended adverse consequences. The current study supports, at a detailed spatial scale, the emerging concern that land-based climate mitigation policies could adversely affect food security. While climate change alone could be beneficial to UK barley production, projected reductions in cropland areas due to land-based climate mitigation policies can combine with increased demand due to population growth to create large deficits in feed barley supply from domestic production. Because feed barley accounts for the largest share of coarse grain used in animal feed, the observed deficits would create correspondingly large deficits in feed barley equivalent meat (FBEM) supply. The situation would be worse if projected median yields are observed. This risk of inadequate supply of meat would have adverse implications for the UK food security as meat is an important source of daily calories, high value protein and essential micro-nutrients. The study demonstrates how a combination of climate change, climate mitigation policies, and population growth can impact on feed barley and meat supply in the future and the need to develop measures to effectively manage the demand and supply sides of feed barley and meat. Because the UK is a strong trading nation and, currently, a major exporter of barley, the prospect of the UK sourcing feed barley or meat from the global market is a case that highlights the potential for high-income countries with high capacity for barley and meat production (such as in the EU) turning to compete intensely with developing countries in the global market. Such intensification of competition on the global market due to supply deficits in major producing countries could make it harder for poor countries to economically access feed barley and/or meat, and therefore adversely affect their food security. It is suggested that a global cooperation around land use is necessary to address the complex issues in the nexus of climate mitigation and food security while paying special attention to the implications for developing economies. This is to ensure that global capacity for certain key crops such as barley is not compromised to exacerbate food insecurity much beyond the net effect of climate change alone or the national scale.

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Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

Appendix 1

See Table 6.

Table 6 Soil hydraulic properties from the SINFO database used in the simulations

Admin. sub-region	Dominant soil	θ_{sat}	θ_{pwp}	θ_{fc}	Rooting depth (m)	θ_{asw} (mm m ⁻¹)
EE	Medium	0.42	0.18	0.33	7	150
EM	Fine	0.49	0.29	0.43	6.8	140
NI	Medium	0.41	0.16	0.31	6.6	150
NEE	Medium	0.42	0.18	0.34	6.6	160
NES	Medium	0.41	0.15	0.30	6.1	150
NWE	Medium	0.43	0.19	0.34	6.4	150
NWS	Medium	0.40	0.15	0.29	7.0	140
SEE	Medium fine	0.55	0.14	0.49	5.9	350
SES	Medium	0.41	0.15	0.32	6.2	170
SWE	Medium fine	0.58	0.15	0.50	4.4	350
SWS	Medium	0.41	0.15	0.31	6.4	160
WA	Medium	0.45	0.22	0.37	6.9	150
WM	Medium	0.45	0.22	0.37	6.7	150
YH	Medium	0.43	0.19	0.35	6.5	160

Data taken from the SINFO database [29]

Appendix 2

See Table 7.

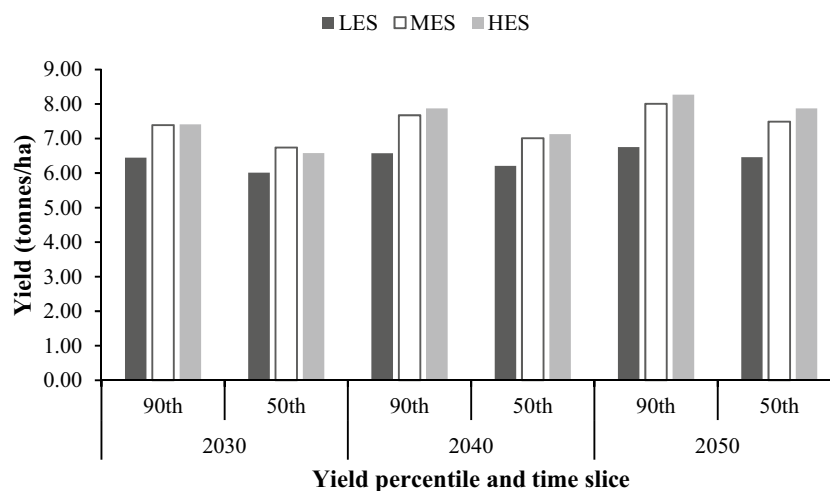
Table 7 Main AquaCrop parameters used in the climate change simulations

Symbol	Parameter description	Value
1. Crop phenology		
1.1. Development of green canopy cover (CC)		
	Initial canopy cover (%)	3.6
	Time from sowing to emergence (GDD)	135
	Canopy growth coefficient (fraction per GDD)	0.8
	Maximum canopy cover (%)	85
	Time from sowing to flowering (GDD)	950
	Length of flowering stage (GDD)	215
	Time from sowing to start of senescence (GDD)	1315
	Canopy decline coefficient (fraction per GDD)	0.06
	Time from sowing to maturity (GDD)	1675
1.2. Development of root zone		
	Minimum effective rooting depth (m)	0.30
	Maximum effective rooting depth (m)	0.70
	Shape factor describing root zone expansion	1.5
2. Crop Transpiration		
	Crop coefficient at maximum CC	1.15
	Decline of crop coefficient (% day ⁻¹) due to ageing	0.15
	Effect of canopy shelter on surface evaporation in late season stage (%)	50
3. Biomass production and yield formation		
3.1. Crop water productivity		
WP*	Water productivity normalized for ET _O and CO ₂ (g m ⁻²)	15
	Water productivity normalized for ET _O and CO ₂ during yield formation (as % WP* before yield formation)	100
3.2. Harvest index (HI)		
	Reference harvest index (Hlo)	0.49
	Upper threshold for water stress during flowering on HI	0.82
	Possible increase (%) of HI due to water stress before flowering	12 (strong)
	Coefficient describing positive effect of restricted vegetative growth during yield formation on HI	Moderate
	Coefficient describing negative effect of stomatal closure during yield formation on HI	Moderate
	Excess of potential fruits	Moderate
	Allowable maximum increase (%) of specified HI	15
4. Stresses		
4.1. Soil water stress		
$P_{exp,lower}$	Lower threshold of water stress for triggering inhibited canopy expansion	0.60
$P_{exp,upper}$	Upper threshold for canopy expansion (canopy expansion ceases)	0.27
	Shape factor for water stress coefficient for canopy expansion	3.5
P_{sto}	Upper threshold for stomata closure	0.60
	Shape factor for water stress coefficient for stomatal control	3.0
P_{sen}	Upper threshold for early senescence due to water stress	0.60
	Shape factor for water stress coefficient for canopy senescence	3.5
P_{pol}	Upper threshold of soil water depletion for failure of pollination	0.80
	Vol% at anaerobic point (with reference to saturation)	15
4.2. Temperature stress		
	Minimum air temperature below which pollination starts to fail (cold stress, °C)	5
	Maximum air temperature above which pollination starts to fail (heat stress, °C)	30
	Minimum growing degrees required for full biomass production (°C day ⁻¹)	15

Appendix 3

See Fig. 8.

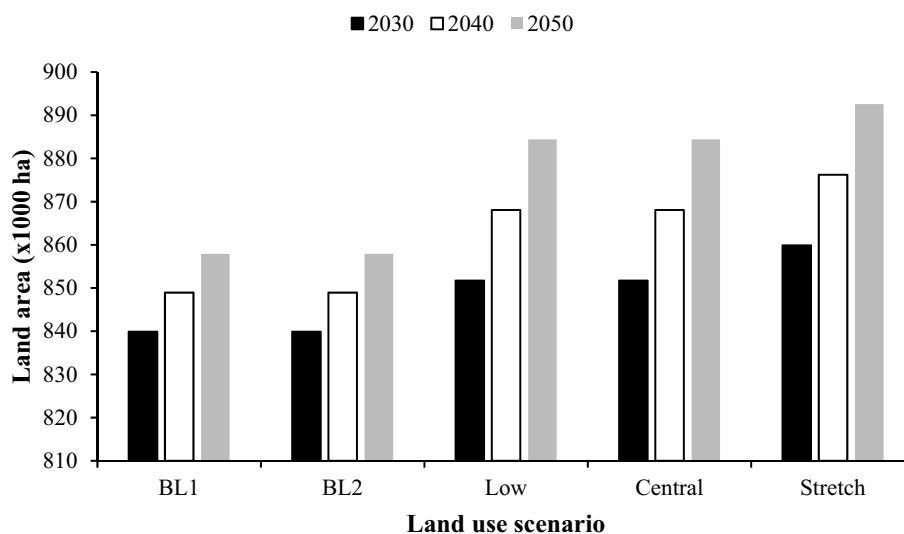
Fig. 8 Projected barley grain yield for the low, medium, high emission scenarios (LES, MES, and HES, respectively) in the 2040s, 2050s, and the 2050s



Appendix 4

See Fig. 9.

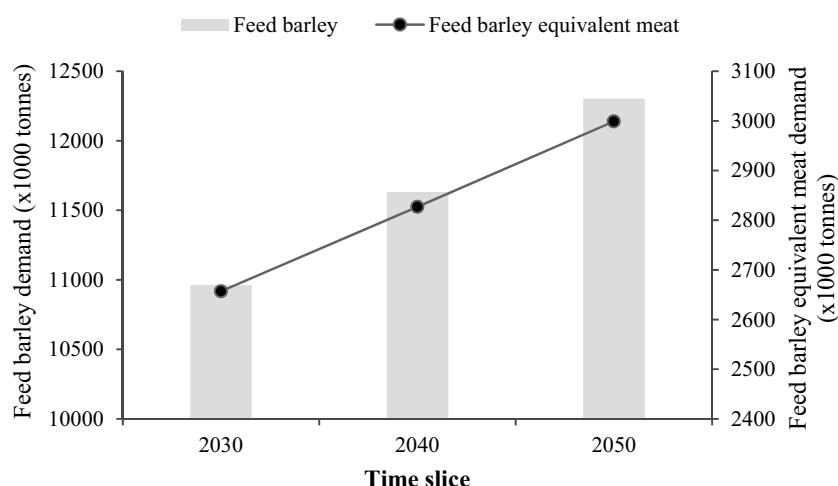
Fig. 9 Projected land area for barley in 2030, 2040, and 2050 based on climate mitigation policies [34]. The total area of barley under the BAU remains constant at 1026 thousand ha from 2030 to 2050



Appendix 5

See Fig. 10.

Fig. 10 Projected feed barley and feed barley equivalent meat demand



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Article

Effect of Climate and Agricultural Land Use Changes on UK Feed Barley Production and Food Security to the 2050s

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Abstract: Currently, the UK has a high self-sufficiency rate in barley production. This paper assessed the effects of projected climate and land use changes on feed barley production and, consequently, on meat supply in the UK from the 2030s to the 2050s. Total barley production under projected land use and climate changes ranged from 4.6 million tons in the 2030s to 9.0 million tons in the 2050s. From these, the projected feed barley supply ranged from approximately 2.3 to 4.6 million tons from the 2030s to the 2050s, respectively. The results indicate that while UK spring barley production will thrive under, and benefit from climate change, total land area allocated to barley production will ultimately determine self-sufficiency. Without expansion in the area of land and/or further significant increases in yields, the UK may face large deficits in domestic feed barley production and, for that matter, meat supply in the future. Hence, agricultural and food security policy needs to consider, principally, the effect of agricultural land use change on key crops, such as barley. Even though the UK can import feed barley or meat to address the deficits observed in this study, the question that needs to be addressed is where all that import will come from.

Keywords: land use change; climate change; feed barley; meat consumption; food security

1. Introduction

The UK Government defines food security as ensuring the availability of, and access to affordable, safe, and nutritious food sufficient for an active lifestyle, for all, at all times [1]. Meat and animal products are a rich source of high value protein and essential micro-nutrients (iron, zinc, and vitamin A), and are therefore key to food security [2]. In this paper, food security is defined as “the risk of adequate food not being available” [3,4]; ‘food’ here refers to meat. Meat contributes a substantial proportion of daily calories in the UK diet.

By 2050, increase in population and incomes will combine with dietary shifts to raise meat consumption per person to about 49 kg for the world and 91 kg in high income countries, with total

meat demand in Europe and Central Asia projected to be 71 million tons [5]. To meet this demand, global meat production should be about 455 million tons [5], which will have a cascading effect on demand for animal feed. Globally, about 35% of total grain produced (the bulk of it being coarse grains) is used for animal feed [6]. Feed use constitutes about 52% and 54% of grains produced in the UK and the EU, respectively [2,7]. As a result of the projected meat demand, global cereal production will need to reach about 3 billion tons by 2050 to satisfy a projected animal feed demand of 1.1 billion tons and other end uses [5]. Feed use of cereal grains is projected to be 35 million tons for Europe and Central Asia [7]. Developing countries could import large quantities of coarse grains as they are projected to account for about 56% of global feed use of coarse grains by 2050 [5].

Globally, barley (*Hordeum vulgare*, L.) is the largest component of coarse grains used as animal feed [4]. By 2050, world barley production will have to increase by 54% over the level in the year 2000 to meet projected demand for food, feed, and industrial purposes [8]. Barley is the most important arable crop in Scotland and the second most important arable crop in the UK. Here, barley accounts for about 39% of feed use of grains and over 60% of total barley produced in the UK is used as animal feed [9]. In 2011, the UK produced 5.5 million tons of barley grains from 970,000 ha of land, with an average yield of 5.7 tons ha⁻¹ [9]. The long-term trend shows that the UK area of land for barley production has remained quite stable (with episodic increases) [9]. Similarly, meat (cattle, pig, chicken, and sheep) production in the UK has not varied substantially from 2005 to 2014 [10]. While the UK has self-sufficiency in barley production, it has a large trade deficit in meat and aggregate animal feed [9]. Recently, the cost of animal feed has become the largest item of expenditure on the agricultural production and income account in the UK due mainly to rising prices of cereals (which increased by 80% between 2005 and 2011) [9]. The cost of feed is projected to remain higher above long-term EU average due to the possible diversion of grains to bioenergy production [11]. At the same time, global demand for animal feed and meat is projected to increase substantially [5,7,8]. As an important source of grain for animal feed, barley plays a crucial role in UK's food security and it is important to assess whether future barley production would be sufficient to meet demand for animal feed. Land use and climate change will be key determinants of the scale of barley production in the future.

"Land use concerns the function or purpose for which the land is used by the local human population and can be defined as the human activities which are directly related to land, making use of its resources or having an impact on them" [12]. Changes in land use arise from competing economic, political, social, and environmental goals. Land use change is a major driver of current global existential and developmental challenges, including climate change, food security, biodiversity conservation, water security, energy sustainability, and poverty alleviation. In the UK, agriculture has the largest share of land use, with cereals (mainly wheat and barley) accounting for over 50% of cultivated lands [13]. Predictions of future agricultural land uses are characterized by great uncertainty [14,15]. Nevertheless, such predictions are useful in providing general conclusions about trajectories of future land use change [14]. Key findings of studies reviewed by Rounsevell and Reay [14] and Angus et al. [16] show: (a) general reductions in the area of croplands and increase in area of land for bioenergy crops and forest; (b) expansion of urban areas with changes in the spatial structure of urban growth and infrastructural networks for land-based transport; and (c) a loss of land in coastal areas. Thus, climate change mitigation and urban development goals will likely be major drivers of land use change and exert enormous pressure on croplands. At the farm scale, market and environmental conditions underpin farmers' decisions, which largely result in crop-specific land use changes [14]. In agro-ecosystems, land use change has direct consequences for the scale of food production and ecosystem services. Thus, the direction (increase or decrease) and intensity (magnitude of change per unit time and land use type) of agricultural land use changes in the UK will have considerable effects on the scale of cereal or barley production.

In northern temperate environments, such as the UK, studies show that projected climate change (changes in the concentration of atmospheric carbon dioxide, temperature, and precipitation) will be beneficial to C₃ cereal crops such as barley through increased photosynthetic capacity and, therefore,

biomass production and yield [17–19]. In addition, it has been reported that, despite potential temperature and water-related stresses, barley production in the UK would remain viable under projected climate change [19,20]. Just like other jurisdictions, studies on UK land use futures [14,16] have focused on aggregate changes in major land use types while climate change studies have focused on impacts on crop yields and risks (e.g., [19,20]). However, the interaction of climate change and land use change will substantially influence the scale of production of specific crops such as barley. Information on the effect of such interactions on cereal production and distribution of cereal crops to end uses for food security is limited. The current study aims to extend the existing literature on the interactive effects of land use and climate change on future food security by assessing the production and the intermediate role of a cereal crop in producing other food. Specifically, this study assesses the combined effect of projected changes in land use and climate change on UK barley production and the consequential effect on feed use of barley for meat production in response to projected demand to the 2050s. The central question being addressed is whether there will be sufficient feed barley supply from domestic production to meet projected demand without adversely affecting other end uses.

2. Materials and Methods

2.1. Current Barley and Meat Indices

The Food Balance Sheet (FBS), published by the Food and Agriculture Organization (FAO) of the United Nations, is widely used for national and international analysis of patterns of food supply and consumption because it is readily accessible and allows international comparisons [21,22]. The FBS provides a picture of the average supply and utilization of food items over a reference period of three years for a given country [21]. A detailed explanation of the components of the FBS is given by the FAO [21]. The FBS is useful for estimating shortages or surpluses of food, projecting future food needs and making policies regarding food production and trade [22]. Barley and meat indices (see Table 1A,B) were retrieved from the UK FBS [23] to represent the baseline. The proportionate domestic use and feed use of barley, as well as the proportionate feed barley in total feed grain, were considered representative for the calculation of future feed barley supply from total production. That is, it was assumed that these proportions would remain unchanged.

In this study, total meat consumption was based on bovine, mutton and goat, and poultry and pig meat. To assess the effect of future feed barley supply on domestic meat production or supply, the current total feed grain was equated to total meat production to allow the calculation of feed barley equivalent meat (FBEM) supply. Obviously, it is an overestimation to equate feed grain use to total meat produced. However, the idea underlying the approach adopted here is that this quantity of feed grain is required or used in the mix of diets to produce the stated total meat. In other words, the idea is to capture the proportional contribution of feed grains to meat production, albeit implicitly, rather than saying that meat is produced only with feed grains. The FBEM (Equation (1)) is, therefore, the quantity of meat (tons) that can be produced or supplied per unit feed barley supplied or consumed (as part of total mix of feed):

$$FBEM \text{ (tons)} = \frac{y}{x} \cdot z \quad (1)$$

where y is the quantity of feed barley in total feed grain (tons); x is the total feed grain (tons) and z is the total domestic meat production (tons). The result of the equation above was divided by the amount of feed barley supply to obtain a constant value that relates unit feed barley required for unit meat production in the future. In this study, feed barley was considered to be used in the production of meat only; the production of milk and eggs was excluded. This was meant to simplify and focus the study. It is true that the production of milk and eggs make substantial use of animal feed made from grains. However, eventually, spent dairy cows and layers are used as meat and thereby contribute to total meat production. Further, different animals for meat production have different feed requirement (in terms of composition and quantity) and feed use efficiencies [24,25]. Again, to simplify the study, it was

assumed that the proportions of feed barley required in the feed of the different animals considered in the current study would remain unchanged in the future.

Table 1. (A) Metrics on barley production and use derived from the UK Food Balance Sheet; and (B) metrics on meat production and use derived from the UK Food Balance Sheet. Data taken from FAOSTAT [23].

(A)		
Item Number	Description	Value
(1)	Total domestic production	5964 thousand tons
(2)	Total export	633 thousand tons
(3)	Total import	115 thousand tons
(4)	Total supplied for domestic use	4953 thousand tons
(5)	% domestic use	(5) = [(4)/(1)] × 100 = 83.0%
(6)	Total supplied for animal feed	3037 thousand tons
(7)	% feed use	(7) = [(6)/(4)] × 100 = 61.3%
(8)	Total supplied for brewing and distilling (considered collectively as 'malt use')	1713 thousand tons
(9)	% malt use	(9) = [(8)/(4)] × 100 = 34.6%
(10)	Self sufficiency	(10) = [(1)/(4)] × 100 = 120.4%
(11)	Per capita barley use	(11) = [(4)/total population] = 80 kg year ⁻¹
(12)	Per capita feed barley	(12) = [(6)/total population] = 49 kg year ⁻¹
(13)	Proportion of feed barley in total feed grain *	38.5%
(14)	Per capita feed grain	153.5 kg year ⁻¹
(B)		
Item Number	Description	Value
(1)	Total domestic production	3.5 million tons
(2)	Total export	695 thousand tons
(3)	Total import	2.4 million tons
(4)	Per capita consumption	84.2 kg year ⁻¹
(5)	kcal supplied per capita per day	457

* Total feed grain (sum of all cereal grain used as animal feed) was represented by wheat and barley used as animal feed since they contributed 96% of all feed use of grains, with oats contributing only 2%. Hence, total feed grain comprised 61.5% wheat and 38.5% barley. Total UK population on the FBS was 61,887,000 people [23].

2.2. The Future Situation

2.2.1. Climate Change and Barley Grain Yields

The Intergovernmental Panel on Climate Change (IPCC) has defined four main families of emission scenarios (A1, A2, B1, and B2; with subdivisions) which are narratives of potential trajectories of greenhouse gas (GHG) emissions as a result of certain assumptions on different socio-economic conditions and development pathways over the course of the 21st century [26]. The UK Climate Projections 2009 (UKCP09) is a publicly accessible online database that provides data on projected climate change (relative to a baseline period of 1961–1990) over the UK [27]. In the UKCP09, three emission scenarios are used: the A1FI, A1B, and B1 (also referred to as high, medium, and low emission scenarios, respectively). Briefly, the A1 narrative represents a future world characterized by very rapid economic growth, rapid availability of new and efficient technologies, fast decline in regional economic disparities and with global population peaking at 8.7 billion in 2050 and declining thereafter to 7.1 billion by the end of this century. The A1 scenario has three subdivisions: A1FI (representing intensive use of fossil fuels), A1T (representing intensive use of non-fossil energy sources), and the A1B (representing an intermediate situation between A1FI and A1T). The B1 narrative portrays a future world inclined towards global equity and sustainable solutions to economic, social and environmental challenges. It also assumes rapid structural shifts towards service and information oriented economies, as well as clean and efficient technologies with less intensive material use. The B1 narrative has the same population scenario as the A1 narrative [26].

Projected daily climate data for the 2030s, 2040s, and the 2050s under the low, medium, and high emissions scenarios (LES, MES, and HES) were obtained from the UKCP09 using the embedded Weather Generator (WG, version 2). Details of the UKCP09 and the WG are provided in Murphy et al. [27] and Jones et al. [28], respectively. The WG randomly samples a specified number of model variants from the probabilistic projections and uses a stochastic process to generate statistically credible daily or hourly future climate variables at 5 km resolution [28]. The generated future climate data, together with data on hydraulic properties of soils obtained from the Crop Growth Monitoring System (CGMS) database in the New Soil Information System (SINFO) used for agricultural monitoring in the European Union [29], and information on growth of the spring barley genotype ‘Westminster’, were used to simulate future barley grain yields for each of the time slices and emissions scenarios. The HGCA Recommended List shows that the genotype Westminster is widely grown in the UK, both as spring and winter barley crop for feed and malt, and is high-yielding. The AquaCrop model [30] was used for the simulations. The AquaCrop model was calibrated using field data from the genotype Westminster grown in 2011 in Dundee (south-east Scotland), information provided in Raes et al. [30], and personal communications with scientists at the James Hutton Institute, Dundee. Further, for the climate change simulations, the ability of AquaCrop to predict barley yields in the baseline period and uncertainties associated with sowing dates were further evaluated by forcing the model (through parameter adjustment) to predict the 1990 yields. To test or validate the goodness of the sowing dates and the model setup, the prediction error of the model was assessed by comparing simulated and observed yields for the baseline period (1980–1989) using the root mean square error (RMSE). With good sowing dates identified, simulations of future yields were done. All simulations were done for rain-fed conditions for the 14 UK administrative regions but the national average is used in this paper. With the exception of Wales, the RMSE values for the baseline simulations for all UK regions were under 1 ton ha⁻¹ [20]. Further details of the simulations can be found in [20].

2.2.2. Land Use Change and Barley Production

The projected areas of UK arable crops for the 2030s, 2040s, and 2050s were taken from Thomson et al. [31]. This was the report of a study conducted for the Department of Energy and Climate Change (DECC). The projections in this report were based mainly on likely policy goals and directions in the nexus of energy, food and climate change [31]. Thomson et al. [31] divided the land use, land use change, forestry (LULUCF) sector into six land use categories: forest land, cropland, grassland, wetlands, settlements, and other land. Four scenarios were constructed (business as usual, low, mid, and high) based on the UK land use policy priorities and aspirations such as targets on food production, reductions in greenhouse gas emissions, or achieving a given extent of forest cover by 2050 [31]. In the business as usual scenario, the current afforestation rate was maintained up to 2050 but other factors were similar to the mid scenario. The mid scenario represented land use change midway between the high and low scenarios. The low scenario focused on the production of bio-energy crops and afforestation. The high scenario prioritized food production with little emphasis on bio-energy crops and forestry [31].

For the current study reported in this paper, future areas of croplands were based on two main scenarios (business as usual and the policy-related land use change represented by the mid scenario provided in [31]), with additional eight scenarios which were adjustment of the policy-related land use change (mid scenario) to represent the effect of non-policy factors on land use change. In the business as usual (BAU) scenario, the average land area for barley production for the period 2000–2012 (1026 million ha) remains unchanged to 2050. The area of land under the BAU scenario constituted 16.36% of the average total area of arable crops for the period considered [32]. This proportion was assumed to remain unchanged to the 2050s and was used to calculate the area of land for barley production from projected cropland areas in Thomson et al. [31] for the three time slices. Thus, the area of land for barley for each future time slice was calculated as a proportion as indicated earlier. The high scenario in Thomson et al. [31] were not used because it significantly increased the area of

croplands and the UK's net greenhouse gas emissions. The low scenario was also not used because of the large increase in the area of bio-fuel crop production and was considered unlikely. Land use futures in the UK will likely be driven by policies sensitive to multifunctional land use and ecosystem services [33], with a balanced focus on energy security, climate change mitigation, food security, and environmental goals aimed at reducing the negative impacts of agriculture [2,16,33].

The remaining eight scenarios were aimed at incorporating possible crop-specific land use changes, in response to market forces and other non-policy factors. To this end, a range of changes from ± 5 to $\pm 20\%$ (at 5% interval), was applied to the land areas under the mid scenario. This range was based on the calculated range of annual changes in the areas of land under barley cultivation over the period 2000–2012 [32], which was -19.41% to 14.94% . To obtain the combined effect of climate change and land use change on future total barley production, the projected areas of land for barley production for each of the time slices and emissions scenarios were multiplied by their corresponding yields of barley from the climate change simulations.

2.2.3. Demand for Feed Barley and Meat

Different reports on projections of global food demand to 2050 exist (see, for example, [5,8,34,35]). In the current study, projected per capita demand for meat and feed grain were obtained from the Comprehensive Assessment of Water Management in Agriculture [34]. This particular study was used because it was the only report which explicitly provided future demand for feed grains. Unfortunately, this report compared future meat and feed grain demand between the OECD (Organization for Economic Co-operation and Development) countries and other regions but not on national basis. However, because it was difficult to obtain the required information for the UK, the projected per capita demand for meat and feed grain for the OECD was used to represent the UK. This has implications for the accuracy of the estimated future demand for meat and feed barley. The good news is that increases in aggregate meat demand in the OECD is expected to result mainly from population growth as changes in per capita meat consumption are projected to be marginal [5,6,25]. The values of meat and feed grain demand for 2025 were used to represent the 2030s and the average of the values for 2025 and 2050 were used to represent the 2040s.

The proportion of feed barley in total feed grain demand per capita was assessed to be 38.5% (see Table 1a) for each time slice. Subsequently, feed barley equivalent meat (FBEM) was calculated (see Equation (1)). Total meat and feed barley demands were then computed as the product of projected population (using the constant population growth scenario data obtained from the UK National Population Projections by the Office of National Statistics) and either per capita meat or feed barley demand. In population projections, the constant fertility scenario, unlike the high and low scenarios, assumes that the fertility rate for 1995–2000 remains unchanged through the future period of interest. The possible supply of barley for domestic uses was assessed to be 83% (see Table 1a) of future UK barley production under each land use scenario, time slice and climate change emission scenario. Similarly, potential malt and feed barley supplies were calculated as proportions of barley supply for domestic uses. Finally, the difference between the projected feed barley or meat demand and supply was calculated to show production deficit or surplus.

3. Results

3.1. Projected Barley Grain Yields

Mean grain yields under projected climate change were highest under the high emissions scenario (HES) in the 2050s and lowest under the low emissions scenario (LES) in the 2030s (Figure 1). For all emissions scenarios and time slices, the magnitude of yield was in the order HES > MES > LES. Differences in yields between the emissions scenarios followed the order 2030s < 2040s < 2050s. Mean yields for all time slices and emissions scenarios were higher than that of the baseline ($4.24 \text{ tons ha}^{-1}$).

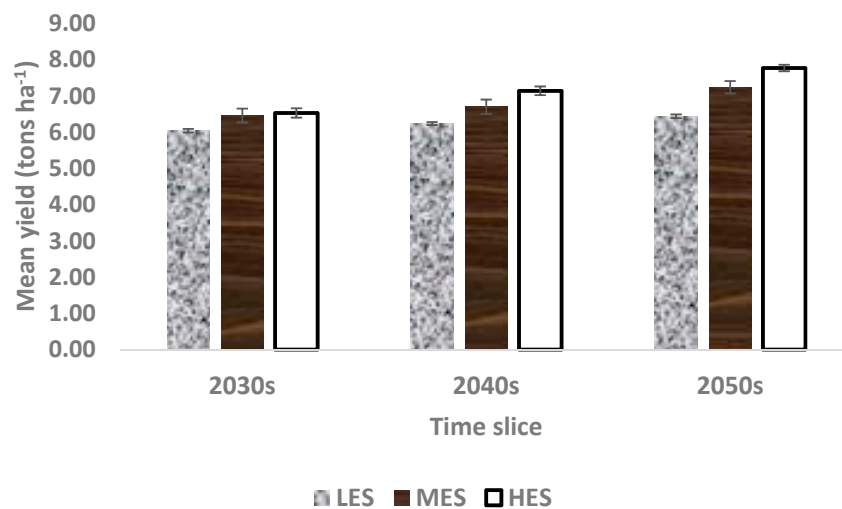


Figure 1. Mean barley yields under the low, medium and high emissions scenarios (LES, MES, and HES, respectively) and three time slices. Error bars are standard errors. Source: authors' simulation results.

3.2. Projected Land Areas and Barley Supply for Domestic Use

For each time slice, the current (BAU) land area for barley production is greater than the projected areas of land under the Mid scenario and the Mid+5% (Table 2). The land areas under the Mid+10% to Mid+20% scenarios are greater than the BAU and all other scenarios. For all scenarios, areas of land for barley production range from 756,000 ha (Mid-20% scenario in the 2030s) to 1156 thousand ha (Mid+20% scenario in the 2050s).

Table 2. Projected areas of land for barley production in the UK. Mid land use scenario data obtained from Thomson et al. [31].

Land Use Scenario	Total Area for Barley Production ('000 ha)		
	2030s	2040s	2050s
BAU	1026	1026	1026
Mid	945	954	963
Mid+5%	992	1002	1011
Mid+10%	1040	1049	1059
Mid+15%	1087	1097	1107
Mid+20%	1134	1145	1156
Mid-5%	898	906	915
Mid-10%	851	859	867
Mid-15%	803	811	819
Mid-20%	756	763	770

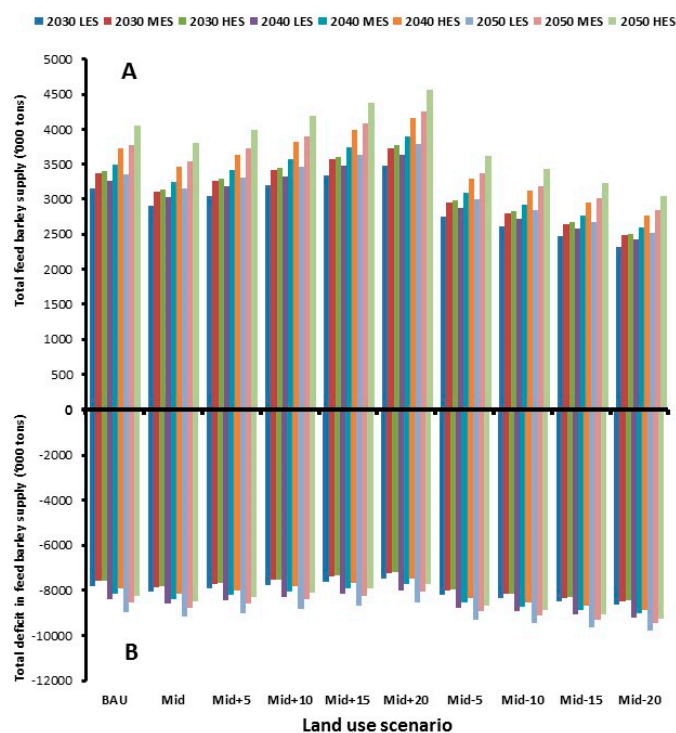
With the effects of climate change on yields, total barley production under the BAU scenario ranged from approximately 6.2 million tons in the 2030s (under the LES) to 8.0 million tons in the 2050s under the HES (Table 3). With the combined effects of projected changes in land area and the climate, projected total UK barley production ranged from approximately 4.6 million tons (under the low emission scenario, LES, and the Mid-20% scenario in the 2030s) to 9.0 million tons in the 2050s under the Mid+20% and the high emission scenario (HES) (Table 3). The difference between the maximum barley production for the BAU and Mid+20%, under the HES in the 2050s, is approximately 1.0 million tons.

Table 3. Projected total UK barley production due to land use and climate change. Data from authors' simulations and calculations.

Land Use Scenario	Total Barley Production ('000 tons)								
	2030			2040			2050		
	LES	MES	HES	LES	MES	HES	LES	MES	HES
BAU	6197	6628	6700	6402	6874	7326	6607	7428	7972
Mid	5708	6105	6171	5953	6392	6812	6202	6972	7483
Mid+5	5993	6410	6479	6251	6711	7152	6512	7321	7857
Mid+10	6279	6715	6788	6548	7031	7493	6822	7669	8231
Mid+15	6564	7020	7096	6846	7351	7833	7132	8018	8605
Mid+20	6849	7326	7405	7144	7670	8174	7442	8367	8979
Mid-5	5422	5799	5862	5655	6072	6471	5892	6624	7108
Mid-10	5137	5494	5554	5358	5753	6130	5582	6275	6734
Mid-15	4852	5189	5245	5060	5433	5790	5271	5926	6360
Mid-20	4566	4884	4937	4762	5113	5449	4961	5578	5986

The potential barley supply for domestic uses shows the same pattern as the total production since the former is a constant proportion of the latter. The largest amount of projected barley supply for domestic uses was approximately 7.5 million tons in the 2050s, under the HES and Mid+20% scenarios; the lowest was 3.8 million tons in the 2030s under the LES and Mid-20% scenarios.

Similarly, projected feed barley supply from domestic production is a constant proportion of the barley supply for domestic use. Feed barley supply under the BAU ranged from approximately 3.2 million tons, in the 2030s to 4.1 million tons in the 2050s (Figure 2A). For the other scenarios, maximum feed barley supply ranges from approximately 3.5 to 4.6 million tons under the Mid+20% scenario, whereas the minimum supply ranges from approximately 2.3 to 3.0 million tons for the Mid-20% scenario.

**Figure 2.** (A) Projected UK feed barley supply from domestic production; and (B) projected deficits in UK feed barley supply from domestic production using the constant fertility scenario of population growth. Data from authors' simulations and calculations.

3.3. Projected Feed Barley and Meat Demand

Using the constant fertility scenario, UK population for the 2030s, 2040, and 2050 was estimated to be 71.9, 76.1, and 80.3 million, respectively (Office of National Statistics). The corresponding total meat demand values were 6902, 7344, and 7789 thousand tons, respectively, while total feed grain demand values were 28,472, 30,212, and 31,959 thousand tons, respectively (data not shown). Similarly, the total feed barley demand (as a proportion of total feed grain demand) was 10,962, 11,632, and 12,304 thousand tons, respectively, for 2030, 2040, and 2050. For the same time slices, the feed barley equivalent meat demand was 2657, 2827, and 2999 thousand tons, respectively.

Consequently, the projected ranges of deficits in UK feed barley supply from domestic production, for all land use and climate change scenarios, were approximately 7194 to 8639 thousand tons for the 2030s; 7473 to 9057 thousand tons for the 2040s; and 7736 to 9623 thousand tons for the 2050s (Figure 2B). The deficits under the BAU scenario ranged from 7553 (under the HES in the 2030s) to 8943 thousand tons (under the LES in the 2050s) while the values for the mid scenario ranged from 7822 (2030s) to 9149 thousand tons in the 2050s.

The corresponding deficits in total meat supply, for all scenarios, ranged from 1744 to 2094 thousand tons in the 2030s, 1816, to 2238 thousand tons in the 2040s, and 1886 to 2384 thousand tons in the 2050s (Figure 3). Under the BAU, the deficits ranged from 1831 (2030s) to 2180 thousand tons in the 2050s. The corresponding values for the mid scenario were 1896 to 2230 thousand tons, respectively.

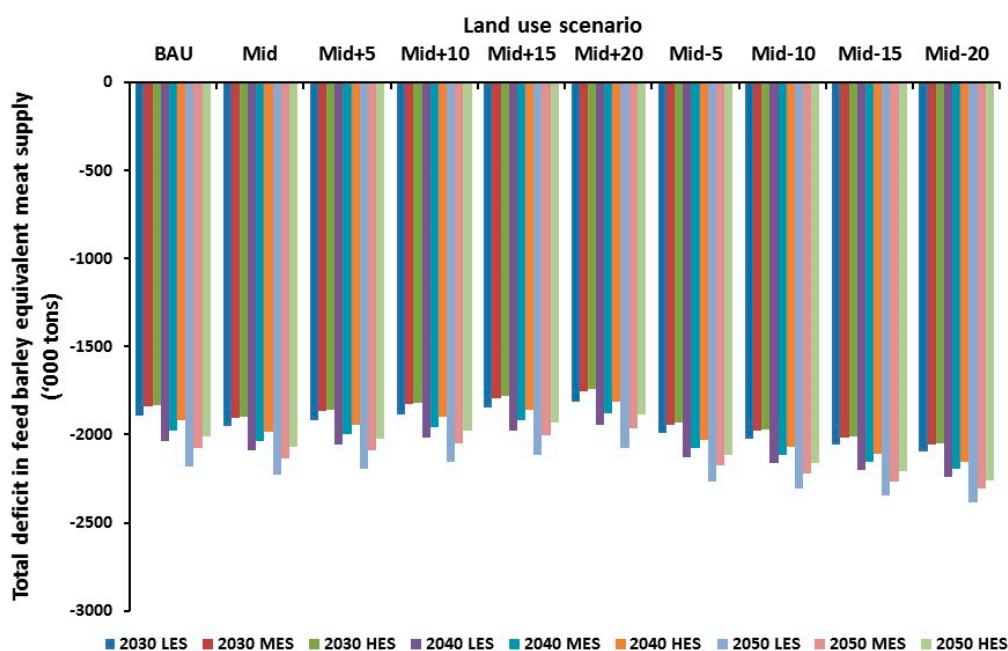


Figure 3. Projected deficits in UK total meat supply due to deficits in feed barley supply using the constant fertility scenario of population growth. Data from authors' simulations and calculations.

4. Discussion

In general, increases in yield will continue to be a preferable path to increasing total production of crops for food security [7]. It is likely that the UK will continue to rely on domestic production as the main source of its feed barley supply. While the UK spring barley production would remain viable under climate change and benefit from yield increases over baseline yields (see Figure 1, [20]), the land area allocated to barley production will be the ultimate determinant of total production. The mid scenario suggests that policy-driven land use change might reduce the area of land for barley

production (relative to the BAU) by as much as 81, 72, and 63 thousand ha by the 2030s, 2040s, and 2050s, respectively. This will result in large reductions in total UK barley production.

In the UK, government policies have been the main driver of agricultural land use change in the past, with factors such as farm incomes, prices and land values playing secondary roles [2,16]. The UK's biofuel obligation under the EU Renewable Energy Directive (2009) and targets on reducing greenhouse gas emissions (as captured in the UK Climate Change Act 2008) might adversely affect agricultural land use futures through a suite of policy and legal instruments, financial and tax incentives, and market signals [36]. Some studies suggest very large areas of land would be required to meet the UK's bioenergy targets up to 2050 [37,38]. For example, the UK might require between 2.7 and 7 Mha of land to satisfy its biofuel needs by 2050 [37]. By the year 2020, biofuel crop production would likely shrink the area of land under barley production by 21% in the EU [11]. Thus, policies regarding climate change and bioenergy production might cause future reductions in the area of food crops and, for that matter, barley.

Similarly, non-policy or market-related factors can cause short term changes in total production through changes in the area of land allocated to barley production [16,39]. The results in the present study show that if these factors increase land area for barley by 20% over the projected level (mid scenario) and under the HES, the absolute increase in total feed barley supply, relative to the BAU, would be 359, 432, and 512 thousand tons for the 2030s, 2040s and 2050s, respectively (Figure 2A). The corresponding reductions in total production under the Mid-20% scenario are 897, 955, and 1010 thousand tons, respectively. Reductions in total barley production could affect the supply of barley for domestic uses or feed.

4.1. Deficits in Feed Barley or Feed Barley Equivalent Meat Supply

Currently, the UK is largely self-sufficient in barley production and almost all its feed barley is supplied from domestic production [9]. Based on the results in the present study, the UK might face large deficits in feed barley or feed barley equivalent meat supply under all land use and climate change scenarios (Figures 2B and 3). Population growth will principally account for increases in meat demand in high-income countries, such as the UK [5,6,25]. Projected increases in population and aggregate meat demand require substantial increases in land area allocated to barley production to offset the potentially large deficits in feed barley or feed barley equivalent meat supply. The difference between future demand and current feed use of barley (Table 1) are 8, 8.6, and 9.3 million tons, respectively, for the 2030s, 2040s, and 2050s. The corresponding feed barley equivalent meat demand for each future time slice is greater than the current quantity of meat imported to the UK. Such large deficits would require integration of policies to ensure a sound balance between domestic production and imports, while addressing the issues of population growth and meat consumption.

It is noteworthy, however, that different animals have different grain feed requirements and feed conversion efficiencies [24,25]. For instance, beef cattle require about five times the dietary energy required by poultry. Thus, the actual total feed barley that will be required in the future will also depend on the proportions of different types of meat that will be demanded. In the UK, a decline in the consumption of carcass meat (mainly beef and lamb) has been observed, while the consumption of non-carcass meat (poultry, pork, and processed meat) is either stable or increasing [9,32]. A similar observation has been made in the EU [11] and the USA [40]. However, this trend might change with improvements in the economy or the socio-economic conditions of consumers [11] due to different price and income elasticities of food components. For example, in the UK, while the average household expenditure on carcass meat is increasing, the average quantity purchased is decreasing [32]. Thus, the observed decline in absolute quantity of carcass meat consumption in the UK might be due to an inflationary effect, possibly through increase in the cost of animal feed [9], rather than a substantial switch to non-carcass meat as substitutes.

It has been suggested that import of meat and animal feed to Europe will increase substantially in the future [7,11]. This might occur through greater attention being given to bioenergy needs to the

detriment of feed use of grains to meet renewable energy targets [7,11,36]. As a result, it has been suggested that barley can lose about 21% of its total area of land in the EU to other biofuel cereals, such as soft wheat and maize [11]. Since the EU remains the largest source of meat and animal feed imports to the UK, the food security question that needs to be addressed is: where will the feed barley or feed barley equivalent meat come from to balance the large deficits observed in the current study? Further, it is important to assess the implications of the UK's exit from the EU for future imports of feed barley or meat to the UK.

4.2. Limitations to the Study

Projections are not equivalent to forecasting or predicting the future, especially over long time horizons for which uncertainties are larger. Rather, projections provide a broad overview of the future state of what is being projected [2]. Hence, projections are not to be considered as prescriptions for the future, but as useful inputs for triggering discussions on alternative adaptive pathways to possible future states of the phenomenon under consideration. This notwithstanding, projections should be based on reasonable analysis of current practice or knowledge. However, finding fit for purpose data for coupled land use and climate change analysis is a challenge [33]. This paper has presented projections of future feed barley supply as affected by land use and climate change and its implications for food security via meat production in the UK. To achieve this, several assumptions had to be made and semi-empirical relationships derived from the current production and feed use of barley as well as meat production or consumption. These have implications for the results obtained in the current study. The proportionality assumption has been central to the current study. For example, one key assumption was that the proportion of feed use of grains (and for that matter barley) in the mix of feed for total meat production based on the animals considered in this paper would remain unchanged to the 2050s. Obviously, policy signals, environmental, market, and other socio-economic conditions can alter the proportional relationships in the future. As a result, the deficits in feed barley or meat production observed in the current study are conservative and applicable only to future conditions that fall within the limits of the current study. For example, the feed barley or meat demand could have been higher or lower, if the high or low fertility scenario of population projection had been used, respectively. In addition, simulations of effects of climate change on crop yields have uncertainties. In the current study, the error associated with the ability of the AquaCrop model to simulate future yields was less than 1 ton ha⁻¹ [20]. However, since error propagation was not done, it is possible that the projected yields could be higher or lower than the observed values used in the current study. Overall, within the limits of the current study, it can be deduced that future reductions in UK croplands would negatively affect feed barley and meat production. This would have a cascading effect on food security via excess demand for meat (due to population growth) over supply from domestic production. This provides a useful input to discussions about future feed use of cereals from domestic production and alternative sources of imports when necessary.

5. Conclusions

Future changes in agricultural land use in the UK will depend largely on the interactive effect of policies regarding food security, climate change and renewable energy. Non-policy factors (mainly market signals) will also affect short term changes in total area of land allocated to a particular crop. Even though climate change can be beneficial to UK barley grain yield, the total area of land allocated to barley will be the key determinant of the UK's future barley production. This study shows that the UK can face large deficits in feed barley or meat supply from domestic production in the future due mainly to inadequate area of land allocation to barley vis-à-vis the large demand due to population changes. As a result, the UK might need to import large quantities of feed barley or meat to balance the projected domestic deficits. Given the projected level of global and EU demand for meat and animal feed, the challenge remains in identifying where the imports to the UK might come from to balance the large deficits predicted in feed barley or feed barley equivalent meat. All things being equal, and

in lieu of large imports, the UK would need to substantially expand the area of barley production to offset potential food security risks. The UK would also need to tightly couple land use and food security policy goals to secure adequate production of key crops, such as barley.

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