Using remote sensing to assess peatland resilience by estimating soil surface moisture and drought recovery

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Abstract

Peatland areas provide a range of ecosystem services, including biodiversity, carbon storage, clean water, and flood mitigation, but many areas of peatland in the UK have been degraded through human land use including drainage. Here, we explore whether remote sensing can be used to monitor peatland resilience to drought. We take resilience to mean the rate at which a system recovers from perturbation; here measured literally as a recovery timescale of a soil surface moisture proxy from drought lowering. Our objectives were (1) to assess the reliability of Sentinel-1 Synthetic Aperture Radar (SAR) backscatter as a proxy for water table depth (WTD); (2) to develop a method using SAR to estimate below-ground (hydrological) resilience of peatlands; (3) to apply the developed method to different sites and consider the links between resilience and land management. Our inferences of WTD from Sentinel-1 SAR data gave results with an average Pearson's correlation of 0.77 when compared to measured WTD values. The 2018 summer drought was used to assess resilience across three different UK peatland areas (Dartmoor, the Peak District, and the Flow Country) by considering the timescale of the soil moisture proxy recovery. Results show clear areas of lower resilience within all three study sites, which often correspond to areas of high drainage and may be particularly vulnerable to increasing drought severity/events under climate change. This method is applicable to monitoring peatland resilience elsewhere over larger scales, and could be used to target restoration work towards the most vulnerable areas.

Key words: Blanket bog, SAR, Sentinel-1, hydrology, water table dynamics

1.Introduction

There is widespread interest in the resilience, or otherwise, of ecosystems subject to global changes, especially climate change (Côté & Darling, 2010). However, despite theoretical progress, there is a paucity of work quantifying the resilience of different real-world ecosystems (Pimm et al., 2019). This is a critical gap, because if we cannot quantitatively measure resilience we cannot tell which ecosystems are losing resilience, and we cannot tell whether efforts to increase the resilience of particular systems are having the desired effect or not. In order to understand resilience it is important to understand the ecosystem response to stressors, but also the response to management techniques used to manage vulnerability (Chambers et al., 2019). The definition of what is meant by resilience can vary across disciplines (Müller et al., 2016); in this study we define resilience as the rate at which a system recovers from perturbation (Chambers et al., 2019; Pimm, 1984; Swindles et al., 2016). This definition of resilience is more easily quantified than definitions of resilience which relate to ecosystem stability over longer timescales, but there is assumed to be a correlation between the two (Hillebrand & Kunze, 2020; Müller et al., 2016; Zelnik et al., 2018). In this study we consider the recovery of water levels following a drought event, an example of a pulse-disturbance, as this is a clearly defined event with a recovery rate that is relatively easy to monitor, and which may give some insights regarding ecosystem stability over the longer term.

Here we take peatlands as a target ecosystem for monitoring resilience. Peatland environments are valuable in terms of the many ecosystem services they provide, including carbon storage (Gorham, 1991), biodiversity (Minayeva et al., 2017), and water purification (Wallage et al., 2006). However, these landscapes are vulnerable to both climate change and human land use pressures (Clark et al., 2010; Gallego-Sala et al., 2010; JNCC, 2011). Given their function as large stores of carbon, there is particular interest in how peatlands will respond to climate change, and how resilient they will be in the face of climatic pressures such as increasing droughts (Page & Baird, 2016). Positive feedbacks include a change in vegetation species towards a community which prefers dryer conditions, and changes in the specific yield of peat causing greater WTD fluctuations (Sherwood et al., 2013; Waddington et al., 2015). Compound disturbances, such as the interactions of drainage, fire, and drought, can cause positive feedbacks to dominate, tipping the peatland over a threshold and into an alternative state (Sherwood et al., 2013; Swindles et al., 2016). These compound disturbances are more likely to occur under climate change (Sherwood et al., 2013; Swindles et al., 2019).

Hydrology is considered a key factor in peatland resilience, as high and stable water levels are associated with healthy peatlands, whilst sustained and/or frequent drought can cause degradation and long-term damage (Kettridge & Waddington, 2014; Sherwood et al., 2013; Swindles et al., 2016; Waddington et al., 2015). Some studies have assessed long-term peatland resilience through paleoecology methods (Lamentowicz et al., 2019; Łuców et al., 2020; Swindles et al., 2016, 2019), whilst others have considered shorter-term resilience through field studies (Holden et al., 2011; Sherwood et al., 2013). Peatlands are generally remarkably resilient over long timescales due to the range of negative feedbacks that can counteract the effects of perturbations to the system. Negative feedbacks include increased surface resistance, altered peat deformation and decomposition characteristics during drought events, which act to bring the water table closer to the surface following a drop in Water Table Depth (WTD), and changes in *Sphagnum* species and behaviour (Kettridge & Waddington, 2014; Page & Baird, 2016; Waddington et al., 2015). An increase in the severity or frequency of droughts due to climate change could, however, cause negative feedback systems to lose their effectiveness (Lowe et al., 2018).

Using remote sensing to assess ecosystem resilience is a fast-growing area of research, as such data can cover large areas over long timescales, with increasingly fine spatial resolution and frequent repeat measurements (e.g. Díaz-Delgado et al., 2002; Li et al., 2014; Washington-Allen et al., 2008). Data collected from satellites are particularly useful in monitoring remote environments where repeat field visits would be difficult and expensive. Remote sensing can provide large scale assessments of spatial variation in resilience, which are the most useful for management decisions (Chambers et al., 2019). However, many remote sensing measures of resilience rely on optical data which can only be obtained under clear sky conditions, and can only detect surface changes (Li et al., 2014). For example, several existing efforts have focused on terrestrial vegetation using the normalised difference vegetation index (NDVI) to monitor the response rate to known perturbations, e.g. fires (Díaz-Delgado et al., 2002), or utilising the inverse relationship between temporal autocorrelation and resilience (Verbesselt et al., 2016). Synthetic Aperture Radar (SAR) is different because it can penetrate below the surface into the top few centimetres of the soil, thereby giving information that would not be available from optical data. Here we explore whether we can use remote sensing derived information to monitor a below-ground (hydrological) measure of ecosystem resilience. SAR backscatter can be used to estimate surface moisture in soils, and has been shown to correlate with WTD time series in peatlands and wetlands (Bechtold et al., 2018; Kasischke et al., 2009; Millard & Richardson, 2018; Schlaffer et al., 2016). SAR also has the advantage over optical data that it can penetrate through cloud, which is a great benefit in the wet climates where peatlands thrive (Babaeian et al., 2019).

The SAR sensor carried on the two Sentinel-1 satellites has great potential for monitoring soil moisture fluctuations due to its frequent return interval, particularly at high latitudes where many peatlands are located, and high spatial resolution (down to 10 m). Not many studies have yet considered the performance of Sentinel-1 SAR due to the relatively recent launch of the paired satellites (April 2014 and April 2016) and therefore the short time series of available images (Asmuß et al., 2019; Huang et al., 2018).

SAR cannot be used to directly compare soil moisture across different sites, as the relationship between SAR backscatter and soil moisture is inconsistent and affected by other factors such as surface topography and vegetation structure (Bechtold et al., 2018; Millard & Richardson, 2018). To counteract this, we developed a method that compares a site against itself, indicating how far above or below average a site is for the time of year, and therefore how long a site takes to recover to its usual moisture level after a drought perturbation. We use the developed method to compare the effects of the extreme 2018 spring/summer drought and heat event (Bastos et al., 2020) to previous and subsequent years. As SAR backscatter functionally estimates soil moisture by penetrating the top few centimetres below the soil surface, but we validated the method using more easily available WTD datasets, the resulting estimates are referred to throughout as a soil surface moisture proxy (this is discussed further in Section 4.1.1.).

Our study therefore makes significant steps forward in several areas. First, we consider the abilities of SAR in general and Sentinel-1 in particular to estimate a soil moisture proxy and WTD dynamics in peatland sites, focusing on sites in Great Britain as a contrast to sites in Germany already considered by Asmuß et al. (2019). Secondly, we develop a method to use SAR estimates of the soil moisture proxy to assess peatland resilience to drought, taking advantage of the unique abilities of SAR to measure below-ground changes, and minimising the effect of confounding factors such as surface topography. We use this method to consider three example peatland sites in Great Britain, focusing on areas where compound disturbances are likely to have decreased resilience. Finally, we discuss the wider implications for peatland monitoring globally.

2. Materials and Methods

2.1. Study sites

This work focuses on the Flow Country in Northern Scotland, the Peak District in the centre of Britain, and Dartmoor in the South-West of England (see Figure 1). These three study sites include a range of blanket bog conditions which characterise peatlands across Britain. The Flow Country is one of the largest expanses of blanket bog in the world, with large parts in good condition with no evidence of human activity. Some areas however were drained and planted for commercial forestry in the 1980s, many of which are now undergoing restoration. Areas of the Peak District peatland have, in contrast, been exposed to different combinations of drainage, managed burns, grazing, and pollution as they are close to several urban centres. Dartmoor is a smaller area of upland peat which has experienced drainage, grazing, peat cutting and erosion.

2.1.1. Water Table Depth (WTD) data

Two peatland WTD datasets were selected to validate the SAR data. The first, from sites across the Forsinard Flows RSPB reserve within the Flow Country, contains nine time series of WTD dynamics from peatland areas undergoing restoration or slope-matched control blanket bog areas. Water tables across the Forsinard Flows reserve were monitored using dipwells. The dipwells comprised auger holes cased with 32 mm polyvinyl chloride (PVC) pipe, with 4 rows lengthwise of 2 mm holes at 5 cm intervals and sealed at the bottom with a 32 mm PVC plug. Dipwells were installed to 1 m below surface. Each dipwell was equipped with an Odyssey capacitance probe (1 m), with the logger body encased in a 30-40 cm section of 40 mm PVC piping secured to the 32 mm dipwell using a reducing coupler. This additional section ensured that the logger placement did not significantly reduce the internal volume of the dipwell. The height of the dipwell relative to the peat surface was measured using 3 manual measurements to mm precision at the start of the monitoring period and checked at subsequent data download site visits. Water level data were captured at 30 min intervals, and for the purpose of the present work, calculated as daily averages to match the remotely sensed data. Each of the time series from the Forsinard Flows reserve dataset used in this study is an average of the data series from at least 3 dipwells within 30 m of each other across a site. A central point was used as the reference for downloading Sentinel-1 data.

The second dataset, from the Moors for the Future partnership, comprises four time series from individual dipwells across the Peak District, using daily averages of hourly observations. The dipwells consisted of a 110 cm PVC pipe sunk to 1 m below the surface, with holes drilled through the sides to allow water movement. In each dipwell a HOBO water level logger was suspended on a wire. At each site a second pressure logger was suspended in a PVC pipe above the surface, with holes drilled to allow movement of air, and the results used to calibrate the water pressure readings.

We were unable to access a WTD dataset covering Dartmoor, but the SAR backscatter method is calibrated to the Forsinard Flows reserve and Moors for the Future partnership datasets combined to minimise the effect of local variations.

The location and site descriptions for all selected time series are given in Table I, and the dipwell locations are shown in Figure 1.

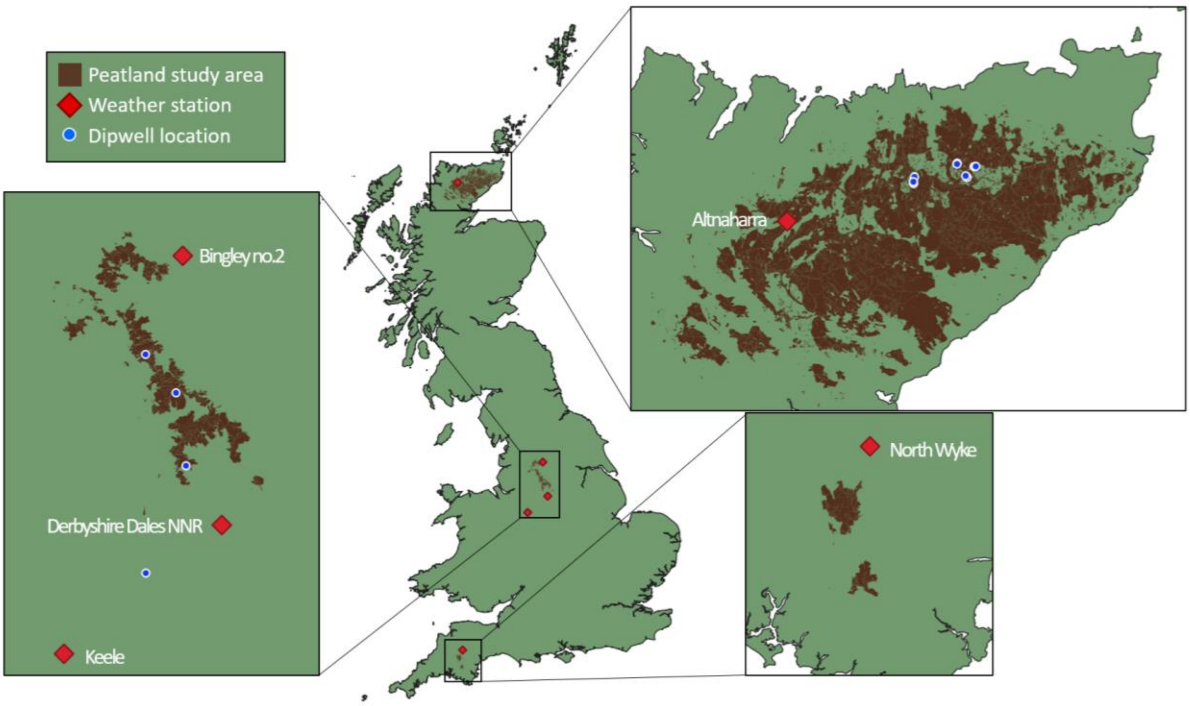


Figure 1 – Locations of the three peatland study areas, weather stations used in SAR backscatter processing, and dipwell locations used to give WTD time series. MftF_R is located on peatland outside of the selected study area.

Table I – WTD time series used in this study. Locations for the sites comprising the Forsinard Flows reserve dataset (ID beginning with F) are central point locations representing the averaged dipwells. Locations with ID beginning with MftF are from the Moors for the Future partnership dataset. Most of the Forsinard Flows reserve sites (except F_CON) were previously planted for commercial forestry, and are currently undergoing restoration starting with tree felling.

ID	**Location (WGS84)**	**Site description**	**Measurements time period**
F_CON	58.3719, -3.9639	Near-natural bog	20/07/2017 – 10/07/2018
F_CL_FTW	58.3853, -3.9612	Felled-to-waste in 2005-6	20/07/2017 – 10/07/2018

F_CL_BCFB	58.3756, -3.9647	Felled-to-waste in 2005-6, brash-crushing and furrow-blocking 2015-16	20/07/2017 – 10/07/2018
F_L_BCFB	58.3875, -3.7601	Felled-to-waste in 2005-6, brash-crushing and furrow-blocking 2017-18	20/07/2017 – 11/07/2018
F_L_FTW	58.3904, -3.7658	Felled-to-waste in 2005-6	18/07/2017 – 25/01/2018
F_R_BCFB	58.4093, -3.7344	Felled-to-waste 2010-11, brash-crushing and furrow-blocking 2014-15	18/07/2017 – 11/07/2018
F_R_FTW	58.4099, -3.7279	Felled-to-waste 2010-11	19/07/2017 – 11/07/2018
F_T_BCFB	58.4152, -3.7995	Felled-to-waste in 1998, brash-crushing and furrow-blocking in 2015-16	21/03/2017 – 12/07/2018
F_T_FTW	58.4135, -3.7998	Felled-to-waste in 1998	22/03/2017 – 12/07/2018
MftF_E	53.3826, -1.8554	Highly degraded with past gullies	09/12/2015 – 19/05/2018
MftF_H	53.5314, -1.8892	*Eriophorum vaginatum* dominated blanket bog, with *Sphagnum* present in small patches	11/09/2015 – 10/01/2018
MftF_M	53.6098, -1.9934	*Molinia* dominated blanket bog with some *Sphagnum.*	09/10/2015 – 01/08/2018

MftF_R	53.1636, -1.9925	Small but relatively intact area of bog. *Eriophorum vaginatum* dominated with *Sphagnum*. This area of bog is in a small dip between two drier slopes dominated (pre-fire) by dwarf shrubs. Fire in August 2018	13/10/2016 – 25/10/2018

2.2. Synthetic Aperture Radar (SAR) data

Sentinel-1 GRD (Ground Range Detected) Interferometric Wide Swath data (spatial resolution of 10 by 10 m), were selected using Google Earth Engine (GEE) (Gorelick et al., 2017). GEE performs thermal noise removal, radiometric calibration, and terrain correction using a Digital Elevation Model on Sentinel-1 data, and completes the conversion from intensity to backscatter coefficient (in decibels, dB). Only images with VV polarisation on a descending pass were used to maintain consistency of imagery.

For comparison with the WTD datasets, the backscatter values of each image were averaged over a circular area of radius 50 m around the point of interest, to reduce speckle (interference) effects.

Processing detailed in sections 2.2.1., 2.2.2., and 2.2.3. was completed for each time series using R (R Core Team, 2017).

2.2.1. Weather filtering

Days with high rainfall (>20 mm) or frozen soil (<2°C) were removed following Bechtold *et al.* (2018), although due to data availability we used soil temperatures at 10 cm depth instead of 5 cm. MIDAS CEDA (Met Office, 2012) datasets from the nearest weather station with appropriate records were used. The station used for the Forsinard Flows reserve dataset was Altnaharra, and the Derbyshire Dales NNR weather station for the Moors for the Future partnership dataset (see Figure 1).

2.2.2. Angle correction

We found a negative correlation between incidence angle and SAR backscatter. This was corrected for by creating an individual linear model describing this relationship for each site, and subtracting this model from the dataset. This correction improved the correlations by an average of 8.6%. Previous studies have found more complicated correction systems give limited improvements in results in wetland environments, and so these were not considered here (Asmuß et al., 2019; Bechtold et al., 2018; Schlaffer et al., 2016).

2.2.3. Sine curve

We found that subtracting a sine curve from the SAR time series for each site improved the correlation with WTD data by a further 48.5% on average. The sine curve was fitted using both the Forsinard Flows reserve and Moors for the Future partnership datasets, and was described by the equation:

$$Y = \sin(0.0173 \times (DoY - 80))$$

Where DoY is Day of Year. This sine curve is a proxy for annual vegetation growth, which can obscure the moisture content backscatter signal. Growing vegetation increases SAR backscatter (Baghdadi et al., 2009), and the sine curve therefore simulates the increase and decrease in backscatter with the growth and senescence of peatland vegetation.

Both the SAR backscatter data and the WTD data for each site were smoothed using the ksmooth function in R, with a bandwidth of 10. Figure 3 presents the SAR datasets resulting from the steps described in Section 2.2. in comparison to the WTD datasets.

2.3. Modelling spatial resilience

GEE was used to select and process the Sentinel-1 datasets as described in this section (see Figure 2). Peatland areas of interest were determined using the EDINA 2015 land cover map (Rowland et al., 2017, see Figure 1).

The SAR data was obtained following the same procedure as described in Section 2.2, filtered for weather conditions as described in section 2.2.1, and corrected for incidence angle as described in section 2.2.2. The sine curve determined in section 2.2.3. was then subtracted. Weather data from Altnaharra weather station was used for the Flow Country, North Wyke for Dartmoor, and Bingley no.2 for the Peak District (as the Derbyshire Dales NNR weather station was not operational after October 2018). Soil temperature data was not available from Bingley no.2 for 2019, so data from the Keele weather station was used to fill in (see Figure 1).

Sentinel-1 tiles covering the area of interest were selected using a central point rather than the entire site polygon, due to the computational limitations of GEE. This means that some of the outlying parts of the study areas have a lower frequency of data points than the central areas.

The 2018 drought was used as a study period, and the non-drought years were separated for creation of an average seasonal cycle. The non-drought years with available Sentinel-1 data (2015 to 2017, and 2019) were used to create an annual average cycle of the soil moisture proxy using the mean of DoY values within 20 days of each date. This average seasonal cycle was subtracted from both the non-drought years and the drought-affected and post-drought period (May 2018 to Dec 2019) to give the residuals. The residuals therefore show whether a date was drier or wetter than the average for that DoY over the non-drought time period. The residuals of May, July, September, and November 2018 were averaged for presentation in Section 3.2 (see Figure 4). Using 2019 data as part of the annual cycle calculation leads to some overlap between the pre-drought and drought-affected data series. This is not ideal, but without the 2019 data included there were not enough DoY data points due to the recent launch dates of the Sentinel-1 satellites. Weather and WTD fluctuations in previous years will also have impacted the annual cycle, and using the maximum data available minimises seasonal anomalies.

The residuals of both the non-drought years and the drought period were smoothed using the average of values within 20 days of each image to minimise the effect of outliers in the data series whilst retaining the trends. The standard deviation of the smoothed residuals of the non-drought period was calculated, and used to detect the starting point of the 2018 drought at each pixel. The drought start point was considered to be the first instance after 1st May 2018 (to avoid anomalies earlier in the year that were not considered part of the study drought) when the residuals fell below the standard deviation of the smoothed non-drought period residuals. The next point when the smoothed residual was greater than the average annual cycle was then found, and the time difference between these two points was calculated to give the length of the drought recovery period at each pixel. If a value greater than the average annual cycle had not been reached after 500 days a value of 500 was assigned.

Results from this method cannot be assumed to be reliable over burnt peatland areas (see Section 4.1 for discussion). Some areas of the Peak District site are regularly burnt, and we have therefore ignored these areas in our analysis (see Figure 8).

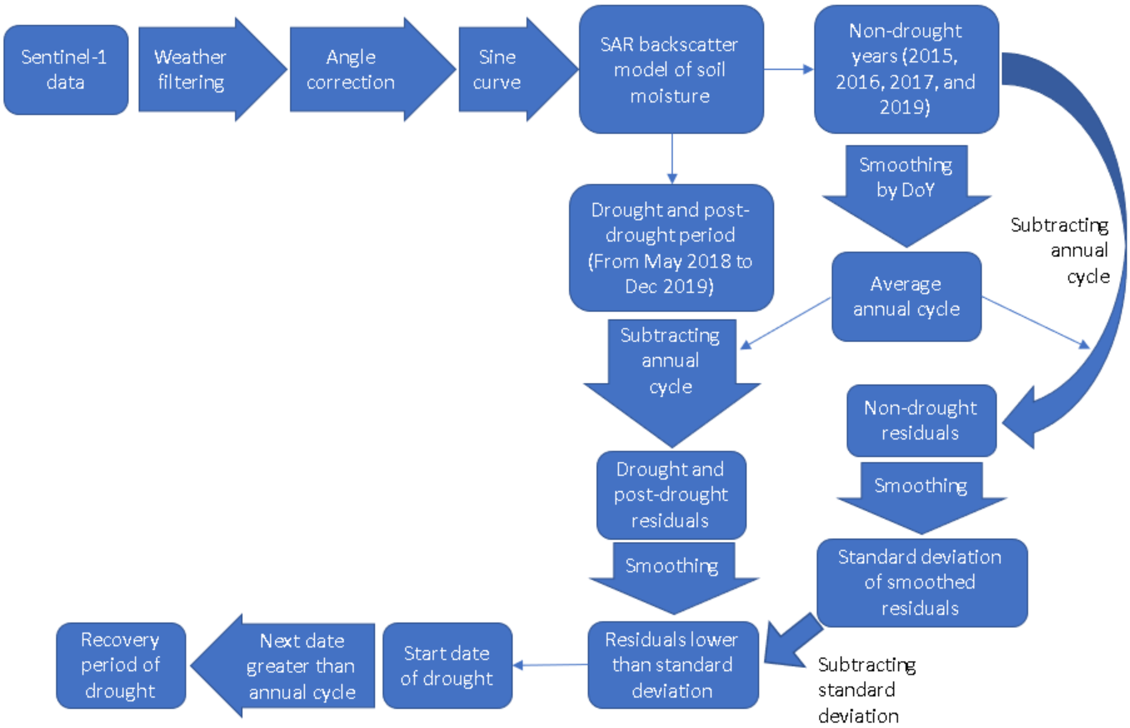


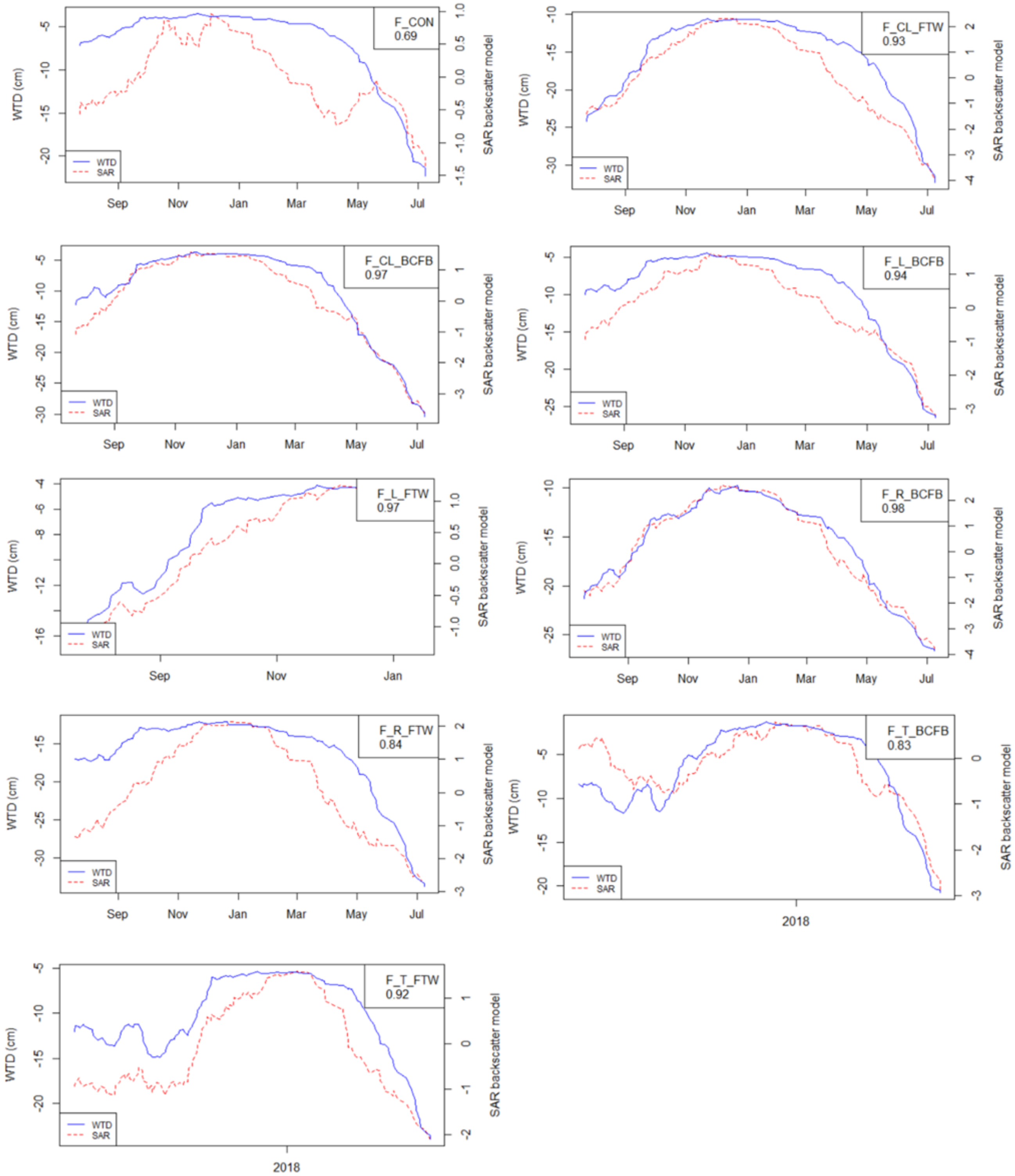
Figure 2 – Flow chart showing the process used to calculate the recovery period.

3. Results

3.1. SAR method

The average Pearson’s correlation between the smoothed WTD dynamics and the smoothed SAR based method was 0.77±0.26 (see Figure 3).

Despite strong correlations for many of the sites, the relationship between WTD and the SAR-based method is not consistent across the selected areas. Some sites, notably F_R_BCFB and F_R_FTW, have higher SAR values in general than the other sites. The MftF_E site has much lower WTD values than the other sites, whilst F_CON and MftF_H have high WTD values across the whole period.


F_CON
0.69
F_CL_FTW
0.93
F_CL_BCFB
0.97
F_L_BCFB
0.94
F_L_FTW
0.97
F_R_BCFB
0.98
F_R_FTW
0.84
F_T_BCFB
0.83
F_T_FTW
0.92
WTD
SAR
WTD (cm)
SAR backscatter model
Sep
Nov
Jan
Mar
May
Jul
2018

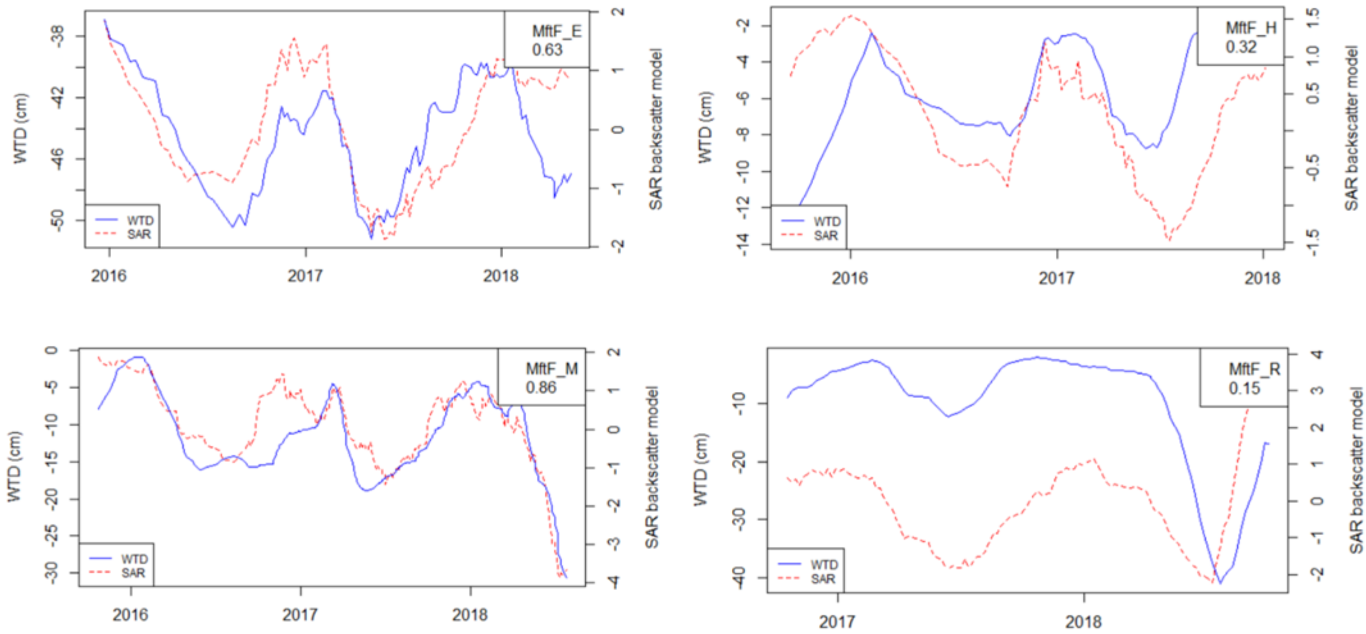


Figure 3 – Smoothed WTD (left-hand axes) and smoothed SAR backscatter (right hand axes) for all sites. All axes are varied to fit the data. Pearson's correlation values are given with the name of the site in the top right corner of each graph.

<u>3.2. Peatland resilience</u>

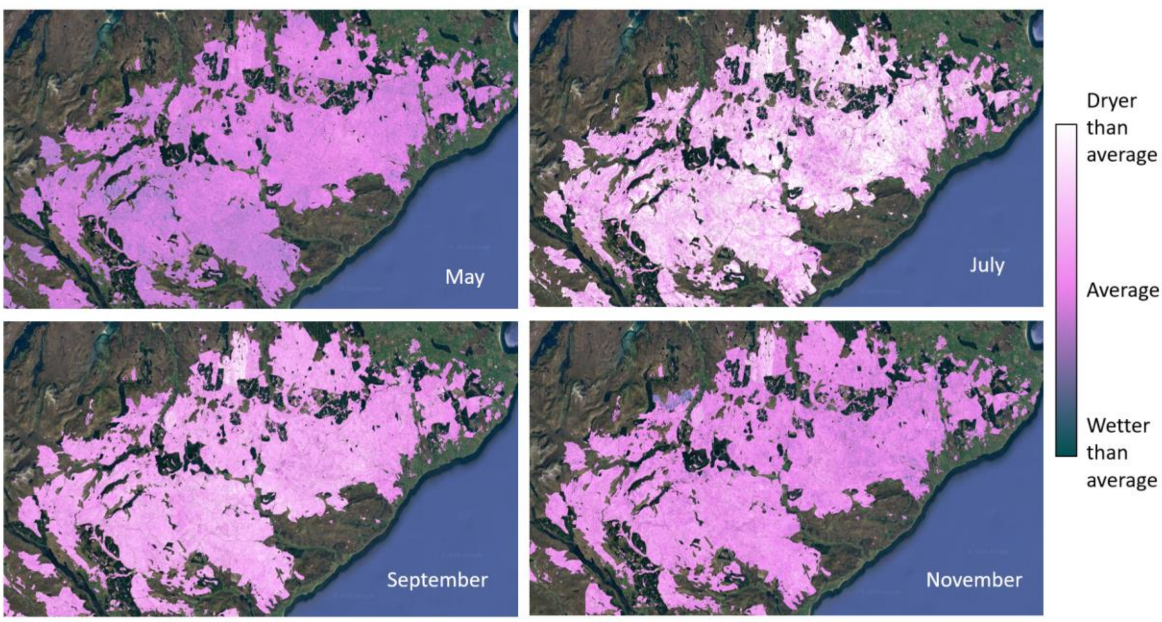
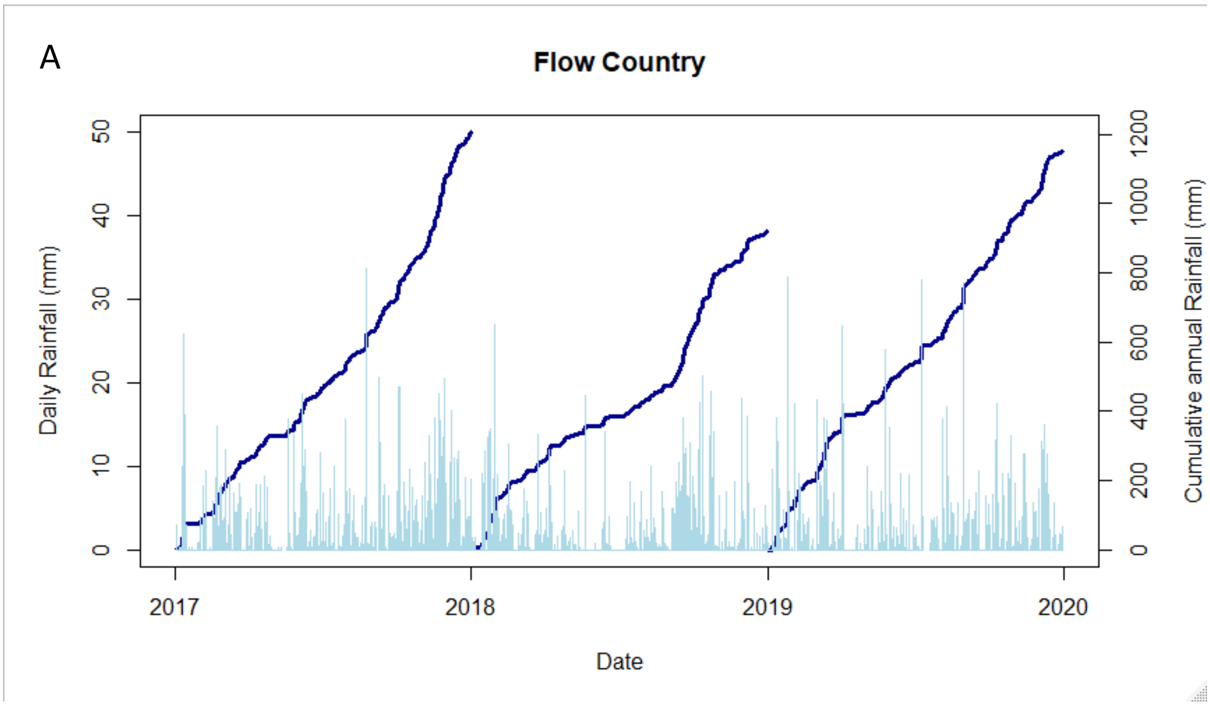


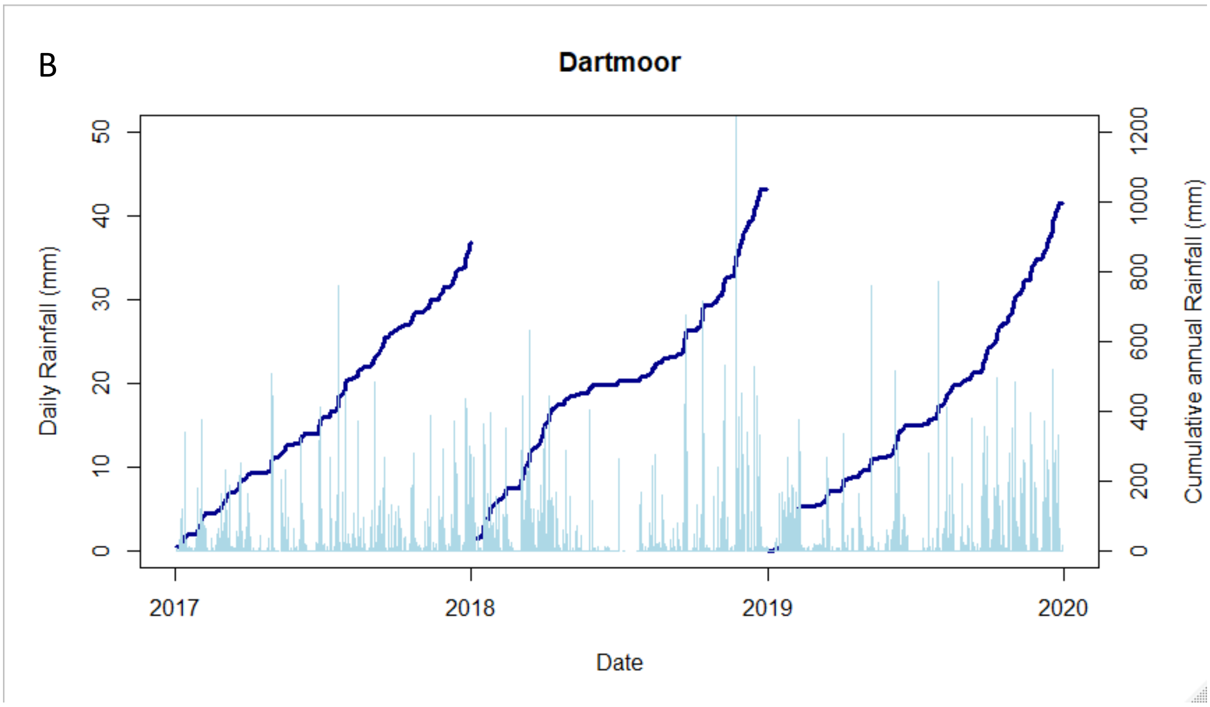
Figure 4 – SAR-based model residuals showing the Flow Country during the 2018 drought period, May to November.

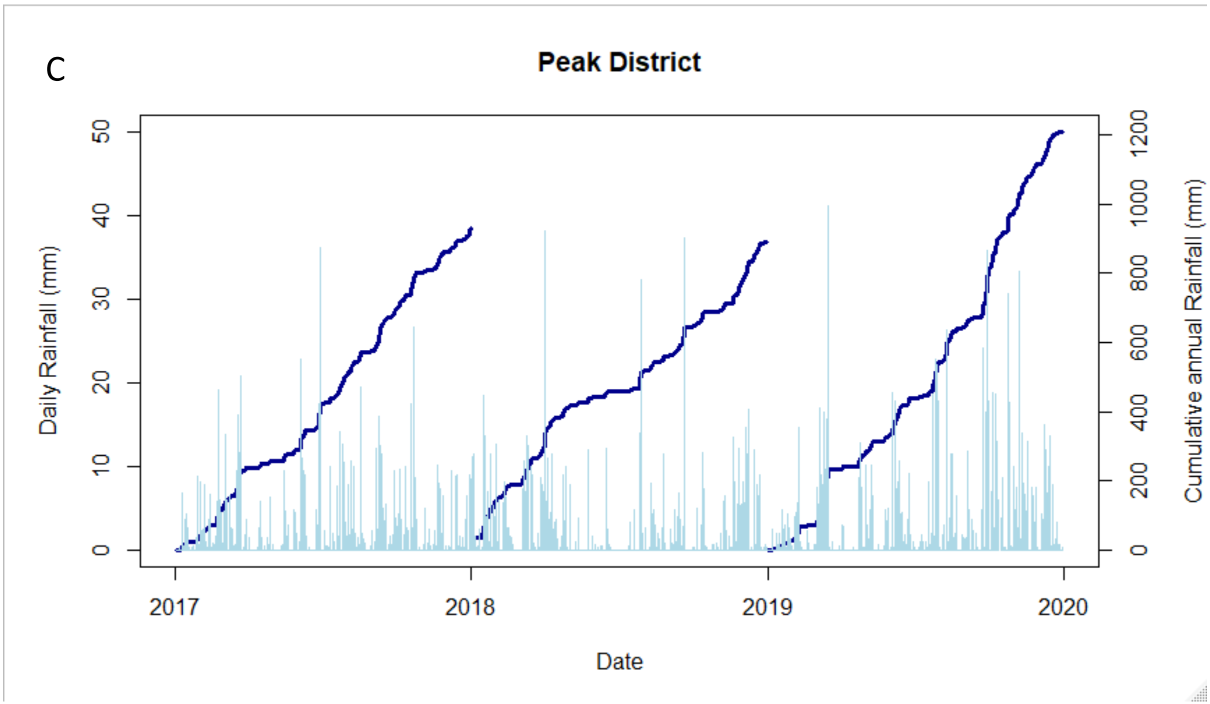
Figure 4 shows the progression of the 2018 drought in the Flow Country peatland area. The site was noticeably drier than previous years in July, and some areas were still drier than average in September and November.

Figure 5 shows daily and cumulative annual rainfall totals for weather stations in the vicinity of the selected peatland areas. Rainfall for the Flow Country (Figure 5A) was lower in 2018 than either the preceding or following year. Dartmoor (Figure 5B), however, shows similar rainfall totals for all three years, although the 2018 drought is evident as a plateau in the cumulative data. The Peak District (Figure 5C) has similar totals in 2017 and 2018, but noticeably more rain in 2019.

Figures 6, 7, and 8 show the length of the recovery period from the 2018 drought in the Flow Country, Dartmoor, and the Peak District respectively. The selected larger-scale sections in these figures show areas of particular interest. Some, such as 6A, 7B, and 8A, show areas that were affected by wildfire in 2018 or 2019. Other areas were selected because they show noticeably longer recovery times than surrounding areas; some of these also have evidence of drainage ditches, such as 6B and C, and others are known to have experienced gullying, such as 8B and C.


A
Flow Country
Daily Rainfall (mm)
0
10
20
30
40
50
Cumulative annual Rainfall (mm)
0
200
400
600
800
1000
1200
2017
2018
2019
2020
Date


B
Dartmoor
Daily Rainfall (mm)
0
10
20
30
40
50
Cumulative annual Rainfall (mm)
0
200
400
600
800
1000
1200
2017
2018
2019
2020
Date

311 

312 *Figure 5 – Rainfall at weather stations in the vicinity of the three selected peatland areas.*

313 *Figure A shows rainfall data from the Altnaharra station, B from North Wyke, and C from*

314 *Bingley no2.*

315

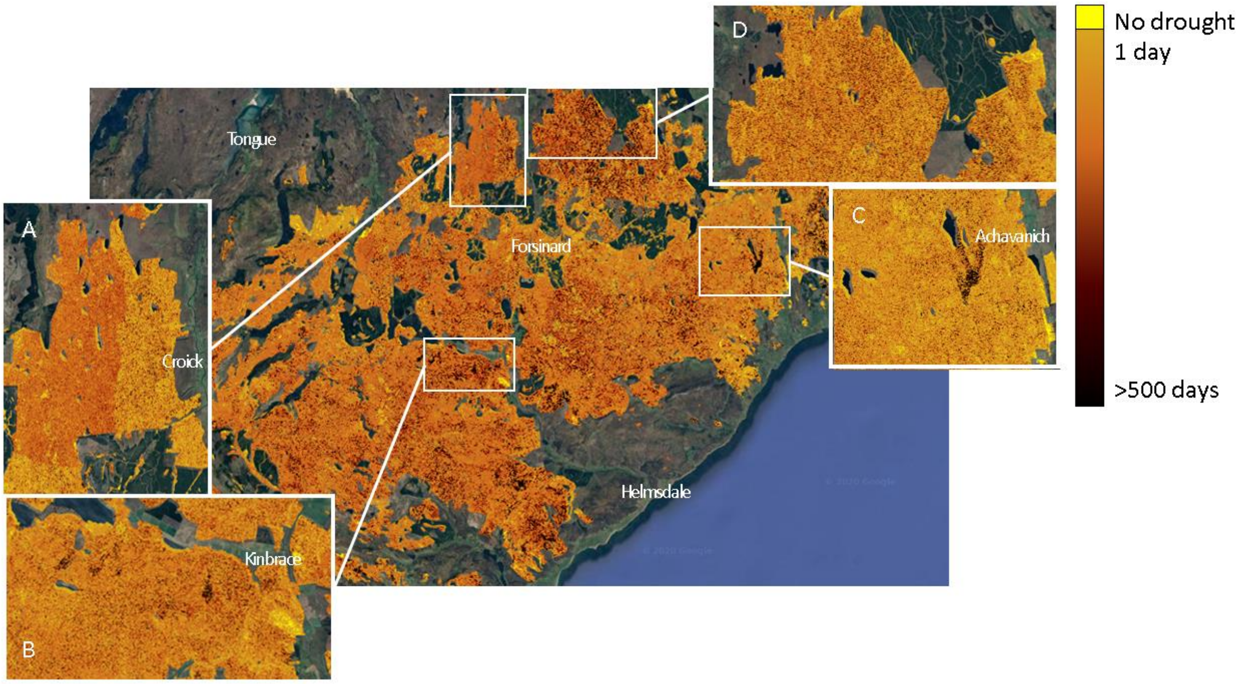
316 

Figure 6 - Length of drought recovery period for the peatland area of the Flow Country. 6A shows the area affected by the Melvich wildfire in May 2019. 6B has a lot of drainage, both natural watercourses and some man-made channels, and 6C has visible drains. 6D shows an area of generally low resilience, although the reasons for this are unclear.

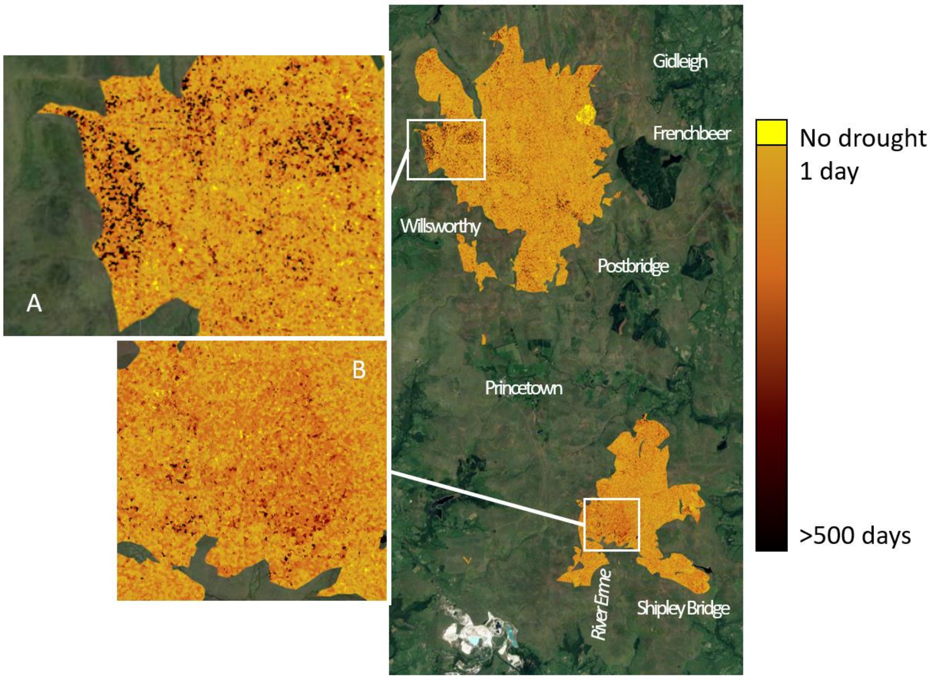


Figure 7 - Length of drought recovery period for the peatland area of Dartmoor. 7A shows an area to the East of Lydford village, which is detected as being less resilient than the surrounding area. 7B shows an area near the source of the River Erme which burnt in a wildfire in April 2019.

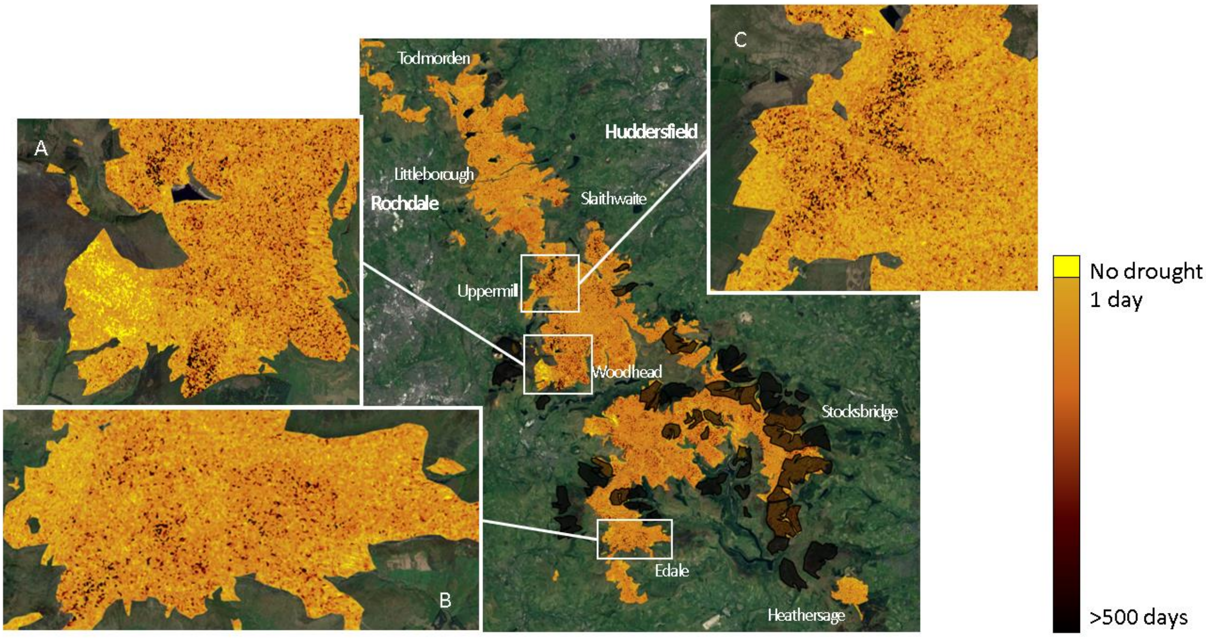


Figure 8 - Length of drought recovery period for the peatland areas of the Peak District. Areas with prescribed burn management have been visually identified and are shown as semi-transparent black polygons; the results should not be considered reliable in these areas. 8A was affected by the Saddleworth wildfire in June 2018. 8B shows the Kinder Scout area, which has extensive gullying, but also much restoration work done over the previous decade (Alderson et al., 2019). 8C also has extensive gullying and had gully-blocking and revegetation work done in 2016

4. Discussion

4.1. Using SAR to estimate WTD

As Bechtold *et al.* (2018) found, SAR backscatter can give reasonable agreement with temporal variation in WTD, but cannot reliably detect spatial differences. This is particularly noticeable for sites F_R_FTW and F_R_BCFB, both of which have large micro-topographical variations due to a relic furrow and ridge system. This micro-topography increases the SAR backscatter values, giving a higher y-axis intersect than other Forsinard Flows reserve sites. The lack of detectable spatial difference is also evident at site MftF_E, which has a low average WTD but similar SAR backscatter values compared to the other Moors for the

Future partnership sites. The method used in this study, which compares anomalies in residuals to the average seasonal cycle of each pixel, allows useful comparisons between areas to be made despite the SAR backscatter data being affected by topography and other factors, as temporal variation in peatland surface roughness is slow compared to soil moisture fluctuations (Millard & Richardson, 2018). Similar to Bechtold *et al.* (2018) we found that the most natural site in the Forsinard Flows reserve dataset (F_CON) had the least agreement with the SAR data, and the most natural Moors for the Future partnership site (MftF_H) also had a relatively low correlation. This may be due to the natural sites being saturated with only minimal fluctuation in WTD for much of the year, or due to the presence of pools (see Section 4.1.1.).

4.1.1. Limitations of using SAR to estimate WTD

Some peatland sites have areas of open water in the form of pools, which have a different relationship with SAR compared to soil moisture and so could disrupt the signal (Kasischke et al., 2009; Schlaffer et al., 2016). Several studies have considered the effect of inundation on SAR backscatter (Kasischke et al., 2009; Schlaffer et al., 2016), and have found that standing water generally reduces the backscatter signal by providing a reflective surface (Bartsch et al., 2012). The peatland sites considered in this study are unlikely to experience complete inundation due to their elevation and topography, but there is standing water present in pools across the Forsinard Flows reserve in the most natural sites, and during winter months in some of the restored sites in the former planting furrows, which may affect the average signal. Many of the pools dried up or at least reduced significantly in size during the peak of the drought, and the resulting fluctuation in SAR backscatter may have affected the recovery estimates.

The results from site MftF_R suggest that fires on peatland can affect the SAR signal in ways which are not yet fully understood. At this site there was a fire in August 2018, and there is a corresponding increase in SAR backscatter following this event. Due to the limited WTD data available from burnt peatland sites, we cannot at this stage say with certainty whether the post-fire changes in WTD shown using our method are reliable, or whether the SAR data is picking up signals from other changes such as vegetation loss. Zhou *et al.* (2019), for example, found that burnt tundra sites had higher backscatter, in line with our results from the MftF_R site. This means that our method should not be applied to sites which are regularly burnt to encourage heather (*Calluna vulgaris*) growth. Some studies have found that WTD is closer to the surface after fire (Clay et al., 2009), whilst others show that it is deeper (Holden et al., 2015). Brown *et al.* (2015) suggest that this disparity may be due to the dominant species on the peatland, with feather moss increasing hydrophobicity and therefore limiting evaporation after fire (Kettridge et al., 2014).

It is important to consider that WTD is an indirect proxy for soil surface moisture, and the accuracy of the model could perhaps be improved if compared against direct measurements of soil moisture, which are unfortunately rarely available (and often inaccurate in very wet peatland soils). It is difficult to calculate soil surface moisture from WTD due to the specific yield, the amount of water needed to raise the water table by a given amount in a given peat volume, which varies across peat types due to the porosity. Where peat is highly decomposed, the porosity and therefore the specific yield is low, leading to large WTD fluctuations (Price, 1996). Capillary processes in *Sphagnum* can also affect the relationship between soil surface moisture and WTD, as can aspect, slope, and vegetation. The WTD datasets used for the method development in this study were mostly only available up to the middle of the 2018 drought, and did not cover the full recovery period. Future work in this area could consider how extreme droughts such as the 2018 event affect the relationship between WTD and soil moisture. It may also be the case that the depth of SAR penetration into the peat is affected by local variations in bulk density, and so the relationship between SAR and soil moisture might also be variable depending on the depth at which soil moisture is measured.

The effects of drainage on WTD may be very localised (Holden et al., 2011) and the speckle (interference) inherent in SAR data may mask these small-scale variations, particularly when using SAR data with a coarse spatial resolution.

4.2. Peatland resilience

Locations where the drought persisted for longer periods of time are understood to be less resilient than areas which recovered faster. These areas may be most vulnerable to the future effects of climate change, notably increasing drought and heatwave events and severities. This method may also be useful as a way of locating areas which could benefit most from peatland restoration, especially where areas of lower resilience correlate with visible drains on the peat surface.

Many of the areas which are shown to be least resilient in the Flow Country and the Peak District have evidence of high drainage, both natural and due to human land management (see Figures 6 and 8).

Both areas highlighted on Dartmoor (Figures 7A and B) have evidence of peatland cuttings, gullies and erosion (Carless et al., 2019), but not noticeably more so than other areas of Dartmoor. Figure 7B, near the source of the River Erme, was affected by wildfire in 2019 (see Section 4.2.1.), whilst the reasons for the longer recovery times seen in Figure 7A may be due to its peatland-edge location. The area to the left of Figure 7A that shows longer drought recovery is an area of high moorland with steep slopes to either side, which may increase drainage. The peat is also potentially thinner there than in the centre of the peat area.

Figures 8B and C show areas which are known to have had extensive gullying, but are now being restored through gully-blocking. Gullying is a form of peatland erosion which can be initiated through the removal of stabilising surface vegetation by fire, pollution, or overgrazing (Evans & Warburton, 2007). Other areas of low resilience may also be affected by erosion, due to management decisions or other factors, that is not visible from satellite imagery.

The area of the Peak District shown in Figure 8C was subject to gully blocking in 2016, and shows a drop in the SAR backscatter signal at around the same time. This may be due to the sudden increase in standing water within the blocked gullies causing a decrease in backscatter due to the reflective surface. The model shows slow recovery in this area, which may be exaggerated due to the seasonal cycle being calculated from both pre- and post-restoration SAR data. The recovery is likely also slow in this area because there would not have been time for the restoration works to be fully effective in recovering peatland function and reversing lower resilience before the 2018 drought. Kinder Scout (Figure 8B), which was badly degraded but had earlier restoration work done (Alderson et al., 2019), shows slow recovery but is not dramatically different from surrounding areas. It may be the case that more recently restored areas have lower resilience because they are not yet in a stable state, and without maintenance and monitoring could again start to degrade. This corresponds with Holden et al. (2011) who found that the hydrology of a peatland area with blocked drains had results between those of a natural and a drained site for several hydrological indicators. Their drain-blocked site had been under restoration for 6-7 years, but the hydrological processes had not yet fully recovered.

Previous studies have found evidence that peatlands become less resilient as compound disturbances increase positive feedbacks (Sherwood et al., 2013; Swindles et al., 2016). Areas which are affected by a combination of anthropogenic disturbances such as drainage and peat cutting, and natural disturbances such as droughts and fires, are likely to be less resilient than areas which have only been subject to one form of disturbance. Our findings show that areas which have been subject to drainage, and areas with severe erosion in the form of gullying, had lower resilience to drought than areas with less evidence of disturbance. This supports the concept of compound disturbances leading to lower resilience.

Increases in the frequency and magnitude of pulse-disturbances can lead to increased variability in ecosystem functions, particularly where resilience is low, and thereby lower stability over the longer term (Zelnik et al., 2018). An increase in drought severity and frequency due to climate change could therefore lead to greater variability in water levels in areas where resilience is low, as the time taken for recovery may become longer than the intervals between drought events. This could lead to a less stable ecosystem and potentially a shift towards an alternative state (Worrall et al., 2006).

4.2.1. Limitations in monitoring peatland resilience through drought recovery

Differing recovery timescales are in part due to varying weather conditions and precipitation. Hence, the three peatland areas considered in this study cannot be directly compared against each other as they experienced different weather conditions during the recovery period (see Figure 5). Recovery patterns within each of the three areas are likely to be more useful for considering local variations in resilience, particularly where there are large differences within a small spatial area. The effects of wildfire in this method are complex. As stated in Section 4.1.1., it appears that fire increases SAR backscatter. This means that the Saddleworth site (Figure 8A) had higher than normal residuals after the fire in June 2018 at the peak of the drought, meaning that the drought effect is not recorded in the model. The areas affected by fire in spring 2019 (Figure 6A in the Flow country and 7B on Dartmoor) show slow but relatively comparable recovery times compared to the surrounding area. The higher post-fire results in 2019 affect the calculated seasonal cycle, thereby potentially exaggerating the effects of the 2018 drought, as can be seen in the Melvich wildfire area of Figure 6A.

Measuring recovery of water levels using SAR backscatter does not give a complete picture of peatland resilience to drought, as there may be chemical and structural changes which occur within the peat during the disturbance event but persist beyond the recovery of water levels (Worrall et al., 2006).

4.4. Future directions

There are several areas of future research which could improve this method and expand its applicability. Considering limitations of the SAR method for estimating WTD identified the variable relationship between WTD and soil surface moisture as a challenge to gaining reliable WTD from SAR backscatter. Future work into the interactions between soil surface moisture and WTD with relation to specific yield would improve our understanding of how these factors are interrelated. Such work should explicitly consider how these relationships change under extreme conditions such as drought. The variations in WTD and SAR backscatter following fire could also be a target for future research. It seems likely that fire causes the SAR backscatter signal to lose correlation with soil moisture and WTD as other factors such as vegetation loss and surface roughness gain more influence (Zhou et al., 2019).

With regards to using SAR backscatter to monitor resilience, future work should consider the perturbation effect size (severity of drought) as well as the recovery time. This would make it possible to compare resilience between peatland areas in different parts of the country and internationally (De Keersmaecker et al., 2015). In the future, SAR backscatter could be considered as part of a range of measures to estimate peatland resilience, some of which would give more information over longer timescales. This would give a greater understanding of the stability of peatland ecosystems under climate change.

4.5. Wider applicability

Using SAR backscatter as a measure of below-ground (hydrological) resilience has the potential to be applied to peatlands, and indeed other ecosystems, around the world. Using SAR to estimate WTD in highly organic soils has already been shown to give good results on sites across Germany (Asmuß et al., 2019; Bechtold et al., 2018), and it is therefore likely that using this method to monitor resilience following drought would also be successful in those ecosystems. In some landscapes, however, other factors affecting SAR backscatter would need to be accounted for. In particular, areas subject to inundation would require the backscatter effects of flooding to be taken into account (Kasischke et al., 2009; Schlaffer et al., 2016). In ecosystems with different vegetation compositions, particularly treed areas, the SAR signal might be obstructed by the canopy, meaning that below-ground measures of resilience may not be achievable using this method (Millard & Richardson, 2018). The use of SAR backscatter is also complicated by the presence of permafrost, which may be a consideration when applying methods such as this to peatlands in the far north (Du et al., 2019). In all cases where this method for estimating resilience could be applied we recommend first validating the relationship of SAR and WTD, or soil moisture if available, with ground data.

In ecosystems where SAR backscatter has been shown to give reliable estimates of WTD/soil surface moisture, there are other applications of this technique besides monitoring resilience following drought recovery. One potential use could be to use a similar method to monitor peatland restoration success, by analysing whether measures such as drain-blocking have been successful in raising water tables across large areas without the need to invest in a significant number of replicate water table monitoring devices.

Although using methods such as this can give useful insights into peatland resilience, the decisions which are made using this understanding are not necessarily straightforward. Chambers et al. (2019) discussed the concept of ‘coerced resilience’, where an ecosystem is maintained by anthropogenic intervention. In such cases, which are likely to become more common due to climate change making certain ecosystems increasingly unstable, resilience is lost and the only way to maintain the preferred state is through human input. In some situations a degraded ecosystem is more resilient than the pristine ecosystem state (Côté & Darling, 2010). This may be the case in some peatland environments where management stressors have already forced the system into an alternative state, which is more resilient to both anthropogenic and climatic disturbances. In such a scenario, decisions must be made as to whether it is worth investing in maintaining or restoring an ecosystem state which may be less resilient to future climate change.

5. Conclusions

In this study we developed a method using Sentinel-1 SAR data to estimate a peatland soil surface moisture proxy, which had an average Pearson's correlation of 0.77 when compared to WTD data. This method was used to derive the residual variation in the soil moisture proxy after the seasonal trend had been removed at three peatland areas in the UK. These residuals were used to assess the severity of the 2018 drought and the time period of recovery across the three sites.

We suggest that the areas which experienced longer recovery periods from the 2018 drought are likely to be more vulnerable to future climate change effects. The results confirmed that there are clear interactions between peatland resilience and human activity, and in particular strong links between peatland drainage and recovery from drought. This supports the theory that compound disturbances weaken peatland resilience.

Our results should be useful for land managers in identifying areas which would benefit most from targeted restoration measures. The method could also in future be adapted to monitor ongoing restoration work. Future work should consider the relationship between peatland fires, soil surface moisture, and SAR data, in order to facilitate remote sensing studies of peatland resilience over areas which are regularly burnt for heather management, or which have experienced wildfire. A stronger understanding of the link between WTD and soil moisture in peatlands would also help to improve the validation of SAR data as a proxy for soil surface moisture, particularly under extreme conditions.

The method developed in this study can be used to provide large scale estimates of peatland resilience from freely available satellite data. This method has the potential to be used to estimate resilience in peatlands across the northern hemisphere, although further validation would be needed on peatland areas with differing vegetation or underlain by permafrost.

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