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Additional Declarations:

Tables 1 to 6 are available in the supplementary files section.

Modelling the hydrological impacts of catchment land use change on a valley peatland in the French Massif Central

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Abstract: The water balance of groundwater- and surface water-fed peatlands depends on inputs from their catchment, which may be impacted by changes in catchment land use. The impacts of alternative land use on peatland hydrological conditions in a small headwater catchment within the granitic uplands of the Massif Central, France are investigated using a coupled MIKE SHE / MIKE 11 model with evapotranspiration modelled using equations based on the HYLUC model. Replacing existing catchment vegetation, dominated by broadleaf woodland, with conifer plantations would reduce overland inflow to the peatland by 50% and groundwater inflow by 14%. Mean stream discharge would decline by 28–47%, with larger impacts in upstream channels compared to downstream of the mire. Discharge during low flow periods would be particularly impacted, with reductions in Q95 ranging from 46% to 82%. Groundwater table (GWT) depth would increase by up to 1 m at some locations, although such large changes would be largely restricted to mire margins and during the driest late summer months. Replacing existing vegetation with grassland or heathland would increase overland inflow by 29% and 22%, respectively. In both cases groundwater inflow would increase by 4%. Mean stream discharge would increase by between 9% and 29% depending on location, and Q95 by between 13% to 89%. In absolute terms, the impacts on GWT depth would be much smaller under the grassland and heathland scenarios compared to the conifer scenario since, under current conditions, the GWT is close to the ground surface for much of the year.

Keywords: wetland; peatland; mire; afforestation; MIKE SHE; Massif Central

INTRODUCTION

Peatlands provide many valuable environmental services (Joosten and Clarke 2002; Parish et al. 2008; Bonn et al. 2016; Tanneberger et al. 2021). In particular, even though they occupy a relatively small proportion of the Earth's land surface, high carbon concentrations in peat make peatlands a globally important carbon store, with a major role in the global carbon cycle and climate system (Lindsay 2010; Froking et al. 2011; Worrall et al. 2011; Yu et al. 2011; Loisel et al. 2021; Strack et al. 2021; Loisel and Gellego-Sala 2022). Mires, i.e. peatlands with active peat-accumulating vegetation, are the most efficient carbon store of all terrestrial ecosystems (Parish et al. 2008). Due to their organic soils and reductive conditions, peatlands can constitute major sinks for almost all metallic and metalloid pollutants, and can efficiently reduce pollutant loads in downstream watercourses (Sobolewski 1999; Brown et al. 2000; Ringqvist et al. 2002; Tipping et al. 2003; Rothwell and Evans 2004; González A et al. 2006; Rothwell et al. 2008; Langner et al. 2012; Lidman et al. 2012; Szkokan-Emilson et al. 2013; Nieminen et al. 2017; Turunen and Sallantausta 2023). In mid-latitudes, mires are relatively rare, providing distinctive environmental conditions and thereby making important contributions to biodiversity. For example, mires cover about 0.14% of the area of Metropolitan France (MEDDE/CGDD/SOeS and Fédération des Conservatoires d'Espaces Naturels 2013), yet 9% of vascular plant species legally protected at the national level, 6% of International Union for Conservation of Nature (IUCN) Red List vascular species and 6% of all vascular plant species are only or principally found in mires (Julve 1996). In recognition of the environmental services they provide, most mire habitats within the European Union (EU) are now protected under the EU 92/43/EEC Habitats Directive that requires member states to designate Special Areas of Conservation (SAC) to contribute to the conservation of a list of habitats and species for which the EU has a global responsibility (European Commission 1992).

The long-term provision of environmental services by mires requires that the water balance of these wetlands is not altered, and that surface peat is maintained at saturation for most of the year (Strakova et al. 2012). Perturbations of mire water balances through drainage, peat extraction, direct afforestation, fire or climate change may lead to a reduction or even inversion in the provision of some services (Strack et al. 2004; Whittington and Price 2006; Sujetovienė and Dabašinskas 2025). A particular concern is the potential for a switch in the radiative forcing of peatlands from negative to positive in response to water level drawdowns (Parish et al. 2008; Harris et al. 2021; Chen et al. 2024). The formation of Dissolved Organic Carbon (DOC) and its release to downstream watercourses have been shown to be promoted by lower and more variable groundwater levels (Freeman et al. 2001; Clark et al. 2005, 2009; Strack et al. 2008; Fenner et al. 2013; Pickard et al. 2022). Similarly, nutrients and metallic pollutants stored in peat can be released into solution and then to watercourses if groundwater levels are lowered or become more variable, for instance following drainage (Rothwell et al. 2009; 2010, 2011; Schöner et al. 2009; Nieminen et al. 2017).

In France, as in other mid-latitude countries, mires are preferentially located in upland and mountainous areas due to relatively high orographic precipitation and lower energy balance and potential evapotranspiration (Manneville et al. 1999; Rydin and Jeglum 2006). About one third of all French mires are within the Massif Central (MEDDE/CGDD/SOeS and Fédération des Conservatoires d'Espaces Naturels 2013), a range of uplands and small mountains in central France. In this area, 79 Special Areas of Conservation have been designated principally or in part for the conservation of mire habitats of European interest under the EU Habitats Directive. The water balance of most of these mires is strongly dependant on surface runoff and groundwater inputs from their wider catchments making them potentially vulnerable to environmental changes beyond their boundaries (Duranel 2015).

Within the Massif Central, in common with many other upland areas in France and around the world (Helmschrot 2006; Mason 2007; He et al. 2014), shifts in the rural economy led to dramatic land use changes over the last century, especially after 1945. There was, in particular, large increases in forest cover, both through active plantation and spontaneous afforestation of land abandoned by farming (Dodane 2009; Derrière et al. 2013). For example, in Nouvelle-Aquitaine, on the western side of the Massif Central, heathlands covered a third of the regional area in the early 19th century, and more than 75% of the landscape on the uplands of the Millevaches Plateau. Since then, 98% of heathlands have disappeared, converted to improved pasture or arable land in the lowlands, and to forests in the uplands. Forests now occupy up to 70% of the landscape in some upland districts (Feydel 2003). According to recent national forestry statistics (Inventaire Forestier National, <http://inventaire-forestier.ign.fr>), 56% of forests on the Millevaches Plateau are coniferous plantations, although this figure masks large local variations. Scots pines, mainly planted in the early 20th century, cover 11% of forested land; Norway spruce 18% and Douglas fir 21%. The latter species, widely introduced after major storms in 1982, is now the species of choice for plantations.

Detrimental impacts of direct afforestation, alone or in association with drainage, on wetlands and their hydrological and ecological integrity is now well known (Jackson 1987; Byrne and Farrell 1997; Vompersky and Sirin 1997; Anderson et al. 2000; Heal et al. 2004; Beheim 2006; Haigh 2006; Oxbrough et al. 2006; Lewis et al. 2013; Limpens et al. 2014; Härkönen et al. 2023; Hughes et al. 2024). At least in some countries, recent regulations and, in some cases, economic considerations have now limited the risks of wetlands, especially large sites, from being directly afforested whilst the viability of afforesting drained peatlands as a climate mitigation measure has been questioned (Jurasinski et al. 2024). However, the knock-on impacts of afforestation within the catchments of wetlands have rarely been investigated. To our knowledge, only a handful of studies have been carried out worldwide (e.g. Smith and Charman 1988; Grenfell et al. 2006; Helmschrot 2006; Krause et al. 2007; Weigandt et al. 2015; Xi et al. 2022). There is, however, strong evidence that in many cases forests are associated with much lower water yields than open habitats (e.g. Andréassian 2004; Farley et al. 2005; Katzensteiner et al. 2011; Alonso et al. 2024; Zhang et al. 2024). For example, Sahin and Hall (1996) analysed results from 145 deforestation experiments carried out across the world and showed that 100% clear-felling would result in an average increase in water yield of 330 mm yr⁻¹ in conifer forests and 169 to 201 mm yr⁻¹ in deciduous forests depending on annual precipitation. Several studies have also reported lower groundwater recharge under conifer forests compared to broadleaf forests, and under broadleaf forests compared to grass, heath or arable land (Cooper 1980; van der Salm et al. 2006, 2007; Rosenqvist et al. 2010; Zhang et al. 2021;

Salem et al. 2023). The main reasons for this are the larger evapotranspiration and interception by trees (Calder 1998, 2003; Hudson et al. 1999; Harsch et al. 2009).

There is palaeo-ecological evidence that large-scale deforestation due to the development of agriculture during the Holocene led to changes in the water balance of many wetlands, and in some cases even triggered the creation of new wetlands. Woodward et al. (2014) carried out a global meta-analysis of 205 wetland sites, including swamps and lakes, where Holocene deforestation had been evidenced and for which it was possible to assess the presence or absence of hydrological changes through palaeo-ecological or historical methods or via direct observation. Increased water inflow following deforestation was identified in 28% of these wetlands, and in 55% of “swamps” (which included all non-lacustrine wetlands). According to a simple water balance model, inflows to 9-12% of wetlands globally, and to 20-40% of Ramsar-designated wetlands, may have increased due to Holocene deforestation. An analysis of the timing of peat inception in the eastern Massif Central also showed that peat formation in a large number of mires, and especially small fens with shallow peat deposits located at relatively low altitudes, could be related to the development of farming which led to both deforestation and the construction of small-scale water management structures (Cubizolle et al. 2012). Reversals of the historic trends of upland forest loss could, therefore, have consequences for peatlands in these areas.

The current study addresses the critical gap in our knowledge of the potential impacts of changes in catchment land use on the hydrology of upland peatlands of conservation significance. This is achieved using a high-resolution MIKE SHE / MIKE 11 coupled hydrological / hydraulic model of a small, but representative, catchment in the western Massif Central that contains a mire with several national and European conservation designations. A particular emphasis is placed on assessing the impacts of afforestation to reflect the land cover changes experienced within the region during much of the 20th century and the first quarter of the 21st century. The hydrological impacts that are assessed include changes in the water balance of the wider catchment and the mire. This focuses, in particular, on the impacts of both overland and groundwater inflows from the catchment to the mire given the importance of these fluxes for sustaining high water tables and saturated peat. In turn, the impacts of catchment land use change on shallow water tables within the mire are also assessed. In this way, the study aims to reinforce the need to consider whole-catchment approaches when formulating peatland conservation policies and practices.

METHODS

Study site

The study site is a 231.3 hectare catchment broadly matching the boundaries of the Duges National Nature Reserve (NNR), located near the small village of Saint-Léger-la-Montagne, in the administrative department of Haute-Vienne, Nouvelle-Aquitaine, France (latitude: 46°00'42"N, longitude: 1°25'07"E, Figure 1). It lies at the heart of the Monts d'Ambazac, a low altitude mountain range at the north-western limit of the Massif Central. The catchment has been described as a prime example of etch-basins (*alvéoles* in French) that are widely found in Hercynian mountains (Valadas, 1998). It comprises a circus-like valley with a flat bottom, surrounded by gentle hills, and opening into a narrow linear valley that, beyond the study catchment's outlet, leads itself to another etch-basin further downstream. Elevation ranges from 532 m above sea level (NGF69) at the catchment outlet to 664 m at the top of Puy de la Garde, on the south-eastern catchment boundary. A 30 m-high residual hill, Puy Rond, rises at the centre of the main basin. The catchment is entirely located on leucogranite, dissected by numerous veins of lamprophyres. According to Valadas (1998), Mauroux et al. (2009) and Duranel (2015), the saprolite layer on top of the leucogranite has been almost entirely removed by erosion and is generally less than 1-2 m deep. The site was designated as a NNR in 1998 mainly to ensure the conservation of a range of acidic mire habitats which cover a total of 43 hectares and are located principally at the bottom of the etch-basin (Durepaire and Guerbaa 2008). It was further designated as a Special Area of Conservation (FR7401135) under the EU Habitats Directive. Within the mire, the average peat depth is 0.80 ± 0.49 m, with a maximum of 3.45 m at its centre (Duranel 2015).

According to Joly et al. (2010), the catchment lies at the transition between an altered oceanic climate and a mountainous climate. Long-term (1981–2010) mean annual precipitation and temperature recorded 4 km away from the site and at an altitude of 629 m asl are 1367 mm and

10.1 °C, respectively. Shorter-term observations (1998–2013) provide corresponding figures for the middle of the mire of 1267 mm and 9.6 °C, respectively (Duranel 2015). Annual total Penman Monteith reference evapotranspiration in the middle of the mire (provided as a standardised estimate of atmospheric evaporative demand independent of vegetation type to enable comparison with sites in different geographical contexts) is 687 mm (Thompson et al. 2026). Precipitation is relatively well distributed throughout the year albeit with a winter peak and summer low whilst reference evapotranspiration is more variable through the year following the reverse seasonal distribution.

The mire's catchment is dominated by semi-natural beech (*Fagus sylvatica*), oak (*Quercus robur*) and chestnut (*Castanea sativa*) woodlands, with some permanent acidic grassland and heathland patches scattered across the site. A few coniferous plantations (mainly Douglas fir *Pseudotsuga menziesii*, with some Scots pines *Pinus sylvestris*) exist but are relatively limited in extent. Some wet woodlands, dominated by willow (*Salix* spp.) and birch (*Betula* spp.), are scattered within the mire and along its margins.

Coupled hydrological / hydraulic modelling

This study uses the coupled MIKE SHE / MIKE 11 hydrological / hydraulic model of the catchment developed by Duranel (2015), described by Duranel et al. (2021), and most recently employed in a comprehensive climate change impact study by Thompson et al. (2026). MIKE SHE is a deterministic, fully distributed model that simulates actual evapotranspiration, unsaturated flow, saturated flow and overland flow processes in an integrated manner (Graham and Butts 2005; Refsgaard et al. 2010). Previous studies have demonstrated the utility of the system for modelling a range of wetlands including investigating alternative scenarios involving climate change, water level management and environmental restoration (e.g. Al-Khudhairy et al. 1999; Thompson 2004; Thompson et al. 2004, 2017, 2023; Hammersmark et al. 2010; Bourgault et al. 2014; House et al. 2016a&b; Clilverd et al. 2016, 2022; Gardner et al. 2019). MIKE SHE originates from the Système Hydrologique Européen (SHE) developed in the late 1970s and 1980s (Abbott et al. 1986a, 1986b). In the early stages of its development, SHE was fully physically-based: each hydrological process was modelled using a set of physical equations describing mass flow and momentum transfer, such as the Darcy equation for saturated flow in porous media, or the Richards equation for unsaturated flow. It was also fully distributed: the governing equations were solved using finite-difference approaches based on a gridded spatial discretisation common to all processes. MIKE SHE has evolved into a suite of modules that also include some simplified process descriptions that, providing assumptions are made and are justified for the system being modelled, allow for the use of simplified physical equations or even conceptual models based on distributed, semi-distributed or lumped approaches (Refsgaard et al. 2010).

As described by Duranel et al. (2021), the Duges model combines physically-based and conceptual modelling methods to reduce computational demand to acceptable run times whilst still providing appropriate representation of key hydrological processes. The empirical two-layer unsaturated zone (UZ) model (Yan and Smith 1994) was preferred to the full Richards (1931) equation. This choice was made principally for computation time reasons, but was also justified by the absence of detailed data on unsaturated zone properties as well as the particular suitability of this method for shallow water table conditions that characterise the mire (e.g. Thompson 2012). The lower number of parameters required by the two-layer model (saturated hydraulic conductivity, volumetric soil moisture at saturation, field capacity and wilting point) facilitates their estimation through calibration. Overland flow was modelled using a 2D, finite-difference, diffusive wave approximation of the Saint-Venant equations, solved using the successive over-relaxation method.

The saturated zone (SZ) was modelled using an iterative implicit finite difference technique to solve the 3-dimensional Darcy equation. A 10 m × 10 m resolution was used, resulting in 23,111 model cells. A no-flow boundary was assumed around most of the boundaries of the topographic catchment. At its downstream end a constant-gradient boundary was used, assuming that the slope of the phreatic surface broadly follows that of the topographic surface. Based on field investigations using geological drilling, electrical resistivity tomography and analysis of available outcrops (Duranel 2015), two saturated zone computational layers were specified. The lower layer corresponds to the fissured granite, with a constant depth of 55 m below the topographic surface. The depth of the upper layer is variable. Within the mire, it follows the boundary between peat deposits and underlying mineral formations, with a minimum depth of 0.5 m to avoid numerical instabilities. On mineral soils,

it was 2 m deep, to match the estimated root depth of woodlands. This is because the root zone in MIKE SHE cannot extend beyond the bottom of the uppermost computational layer, whatever method is used to model evapotranspiration and unsaturated flow. When soils and superficial formations are relatively shallow, as in the Dauges catchment, a trade-off was made between appropriate representation of root depths, an essential factor driving differences in evapotranspiration between vegetation types, and an accurate representation of the physical properties of the material within the root zone. Here the uppermost computational layer on mineral soils corresponds to a complex of soil, peri-glacial formations, in-situ saprolite and, in a substantial part of the catchment, the top of the fissured granite layer.

The MIKE SHE model was coupled to a MIKE 11 1D river model of the main channels within the catchment. To keep run times within acceptable limits, channel flow was modelled using a kinematic routing method, and water levels estimated a posteriori based on the Manning's equation. This method is quick, and generally relatively accurate for fast-flowing streams where no backwater occurs. It is not expected to give very accurate representations of stream stage, and consequently it was assumed that exchanges between the overland component and watercourses occur towards the latter only, i.e. there is no over-bank flooding. This is acceptable since within the Dauges catchment over-bank flooding only occurs on a very narrow band along the main stream and in a limited area in the downstream part of the mire (see Duranel et al. 2021). The Dauges MIKE SHE / MIKE 11 model used an adaptive time-step, whilst results were aggregated to a daily resolution.

Modelling evapotranspiration

Since the aim of this study is to model the impact of catchment land use change on the hydrology of the mire, an accurate representation of evapotranspiration processes is required. For this reason, the approach used to model evapotranspiration is further detailed herein. Within MIKE SHE actual evapotranspiration (ET) is estimated in each model time-step by emptying a series of reservoirs in turn: snow storage (assumed to be zero in this study), interception storage, ponded water, the unsaturated zone and, finally, the saturated zone (provided the water table is above the depth defined by the sum of the root depth and capillary fringe thickness). When using the two-layer unsaturated zone approach in MIKE SHE version 2009 (employed herein), there is no distinction between evaporation and transpiration and ET is capped to reference evapotranspiration ET_{ref} multiplied by a crop coefficient K_c . This makes modelling evapotranspiration difficult when evaporation is large and proceeds at a substantially greater rate than transpiration, for instance in forests (Calder 1998; Granier et al. 1999). Furthermore, ET_{ref} is generally calculated using a grass reference (Allen et al. 1998). Therefore, capping actual evapotranspiration to an estimation of maximum evapotranspiration based on ET_{ref} multiplied by a fixed coefficient (1.2 in Granier et al. 1999, for instance) may result in the under-estimation of evaporation from interception in forests, particularly in winter (Calder 2003). Indeed, the Penman-Monteith equation predicts that when the aerodynamic resistance is an order of magnitude lower than the canopy resistance, as is the case in forests, evaporation from interception proceeds at 3 to 5 times the transpiration rate under similar weather conditions (David et al. 2006).

In view of these limitations, a different evapotranspiration modelling approach to those directly available within MIKE SHE was employed with alternative methods being specified in tall and short vegetation types. The original distribution of vegetation within the Dauges catchment was based on a map of CORINE biotope vegetation classes (Commission of the European Communities 1991) reclassified to nine categories (needle-leaved evergreen woodlands, broad-leaved deciduous woodlands, mixed woodlands, heathland, clearings and shrubs, meadows and pastures, wet woodlands, wetland, impervious areas). The approach employed to model evapotranspiration in areas of different vegetation was based on the limits concept developed by Calder (1998). As proposed by Calder (2003), a distinction was made for tall vegetation (woodland, heathland and shrubs) between wet-time evaporation (corresponding to evaporation from interception) and dry-time evapotranspiration (which includes transpiration and any other evaporation from the ground surface). Evaporation from interception was modelled outside MIKE SHE using the same equations as those implemented in the HYLUC model developed by Calder (2003), and widely used to evaluate the contrasting impacts of forests and short vegetation on water resources (Haria and Price 2000; Jewitt et al. 2004; Green et al. 2006; Calder et al. 2009; Delgado et al. 2010). Daily evaporation from interception (I) was calculated as follows:

$$I = \gamma(1 - e^{-\delta P}) \quad (1)$$

where γ and δ are calibrated, empirical parameters, and P is daily rainfall. As proposed by Calder (2003), Eq. (1) was further simplified by assuming that:

$$\delta = 1/\gamma \quad (2)$$

In HYLUC, dry time evapotranspiration (ET_{dry}) is modelled as:

$$ET_{dry} = \beta m E_T (1 - w) \quad (3)$$

The β coefficient accounts for differences in transpiration between different vegetation types in the absence of soil moisture stress and is somewhat analogous to the basal crop coefficient K_{cb} in the FAO dual coefficient method (Allen et al. 1998). Reduction in transpiration under water stress conditions is accounted for by a soil moisture regulation function m . E_T is Penman (1948) potential evapotranspiration for short grass, used as the reference evapotranspiration in HYLUC. Finally, w is the fraction of the day when the canopy is wet and not able to transpire, It is estimated from daily precipitation and the δ parameter:

$$w = 1 - e^{-\delta P} \quad (4)$$

In the current study, dry time evapotranspiration was modelled in a similar way within MIKE SHE, by specifying as the “potential evapotranspiration” a crop-specific dry time potential evapotranspiration calculated externally as:

$$ET_{pot\ dry} = \beta E_o (1 - w) \quad (5)$$

where E_o is the FAO Penman-Monteith reference evapotranspiration for grass (Allen et al. 1998). The reduction of transpiration under limiting soil moisture conditions was accounted for by the two-layer unsaturated flow component within MIKE SHE. Similarly, the “precipitation” rate specified in MIKE SHE was actually the throughfall rate, after interception modelled using Eq. (1) was subtracted from precipitation. The simulated actual evapotranspiration was post-processed to include the interception rate calculated externally.

As recommended by Calder (2003), no distinction was made between wet time evaporation and dry time evapotranspiration for short vegetation types (grassland and wetland). This is justified since the Penman-Monteith equation predicts that evaporation from interception and transpiration occur at similar rates when aerodynamic and canopy resistances are of the same order, which is the case in short vegetation (David et al. 2006). For short vegetation types, the precipitation rate specified in MIKE SHE was therefore the actual precipitation rate, and the potential evapotranspiration the FAO Penman-Monteith reference evapotranspiration multiplied by a crop coefficient. In grassland, this coefficient was assumed to be 0.9 in summer and 0.6 in winter, based on values measured by Janssen (1994, in Spieksma et al. 1996) in oligotrophic acidic grassland on sandy soils in the absence of water stress. In the wetland, a value of 1.05 was used, corresponding to the FAO-recommendation for open water (Allen et al. 1998). Several studies have measured ET/ET_{ref} ratios close to unity in acidic fens and bogs, in particular those dominated by *Molinia caerulea* (Lafleur 1990; Sijsma and Veldhuizen 1992, in Spieksma et al. 1996; Schouwenaars 1993; Kim and Verma 1996; Wastiaux 2000). Since most of these studies used the Penman (1948) open water evaporation as ET_{ref} , this corresponds to a crop coefficient of approximately 1.05 when using the FAO Penman-Monteith method to calculate reference evapotranspiration. Others have shown that the ET/ET_{ref} ratio from *Sphagnum*-dominated bogs can be substantially lower (Kellner 2001 - around 0.50-0.75; Koerselman and Beltman 1988; Lafleur and Roulet 1992; Wastiaux 2000; Lafleur et al. 2005), particularly when the capitulum dries out. In a degraded bog in the Netherlands, Schouwenaars (1990, 1993, cited by Spieksma et al. 1996) showed that the actual

evapotranspiration of *Sphagnum* spp. equalled Penman open water reference evaporation when the water table was within 10–15 cm of the ground surface, but dropped when the water table fell below this critical level. Actual evapotranspiration by *Molinia caerulea* was maintained at around the reference evaporation rate for much longer periods under dry conditions. In the Dauges model, evapotranspiration in wetland areas was therefore assumed to be driven by vascular plants, in particular by *Molinia caerulea*, when they are dominant in summer, but by *Sphagnum* mosses in winter when *M. caerulea* leaves are senescent and reduced in extent due to late summer and autumn grazing. This seasonality was modelled by adopting a crop coefficient of 1.05 at all times and defining the maximum root depth as 60 cm in summer and 15 cm in winter (see below).

In the case of tall vegetation, documented values that could be used to estimate β are virtually non-existent. A review of available studies is presented in Supplementary Materials Table 1. Values are very variable due to differences in crop coefficient definition, the methods used to measure actual evapotranspiration or transpiration, the reference evapotranspiration used and forest stand characteristics. Very few studies have compared transpiration in trees in the absence of water stress with reference evapotranspiration. The method proposed by Allen et al. (1998) and Allen and Pereira (2009) was therefore used to estimate β based on climate, vegetation height, leaf area index and mean leaf resistance. The basal crop coefficient K_{cb} (equivalent to Calder's β) can be estimated as:

$$K_{cb} = K_{c\ min} + K_d(K_{cb\ full} - K_{c\ min}) \quad (6)$$

where $K_{c\ min}$ is the minimum crop coefficient when the soil is bare (about 0.15), K_d is a vegetation density coefficient and $K_{cb\ full}$ is the estimated basal crop coefficient during peak plant growth with full ground cover. K_d can be estimated from ground cover or from leaf area index (LAI):

$$K_d = 1 - e^{-0.7LAI} \quad (7)$$

$K_{cb\ full}$ is estimated as a function of plant height and climate:

$$K_{cb\ full} = F_r \left(\min(1.0 + 0.1h, 1.2) + [0.04(u_2 - 2) - 0.004(RH_{min} - 45)] \left(\frac{h}{3} \right)^{0.3} \right) \quad (8)$$

where h is the vegetation height, u_2 and RH_{min} are the average wind speed and minimum relative humidity at 2 m during the particular growth stage, respectively. The F_r parameter allows for a reduction in the crop coefficient if the vegetation has a higher degree of stomatal control than the reference grass crop. It is given by:

$$F_r \approx \frac{\Delta + \gamma(1 + 0.34u_2)}{\Delta + \gamma \left(1 + 0.34u_2 \frac{r_l}{100} \right)} \quad (9)$$

where r_l is the mean leaf resistance, Δ is the slope of the saturation vapour pressure versus air temperature curve and γ is the psychrometric constant. The values used to estimate β ($=K_{cb}$) for each tall vegetation type are provided in Table 1. LAI and mean leaf resistance, as well as root depth, were estimated using values from the literature for similar vegetation types in similar geographical contexts (see Tables 2, 3 and 4 in Supplementary Materials). For woodlands, vegetation height was estimated based on canopy height measured as part of the National Forest Inventory (<http://inventaire-forestier.ign.fr>) in plots with similar dominant species in the same eco-region (*Plateaux limousins*) as the Dauges catchment to ensure similarity in climatic and edaphic conditions. In the case of wet woodlands, only plots on hydromorphic soils were included, but the sampling area was extended to the nearby *Plateaux granitiques ouest du Massif central* eco-region to compensate for the small number of records available. Mature woodland canopy height was estimated for each woodland type by averaging records obtained from large trees (at least 37.5 cm in diameter) only, except for *Salix* spp. and *Betula* spp. For non-woodland vegetation classes, canopy height was estimated from informal field observations and published values (Rameau et al. 1994).

To parameterise the interception model, a literature survey of bulk annual or seasonal interception ratios, maximum interception capacities or γ values (for studies using the HYLUC model) recorded in similar vegetation types and climatic conditions to those of the Dauges catchment was undertaken (Tables 5 and 6 in Supplementary Materials). These studies suggest a relatively constant annual interception fraction in conifers with a similar architecture, independent of the location and annual precipitation provided the latter is higher than around 700-800 mm (Figure 1 in Calder 2003). In contrast, values for the maximum interception capacity in general and for the HYLUC γ parameter in particular are much more variable. This may result from different methodologies when measuring the maximum interception capacity, or equifinality issues when calibrating the γ , δ and β parameters of the HYLUC model. When, as in the present study, no interception or soil moisture data are available, it may therefore be preferable to calibrate the γ parameter against expected long-term interception ratios, an approach adopted by Delgado et al. (2010) to model the hydrological response of a Mediterranean basin to changes in land cover. In the present study, the γ parameter was calibrated against the mean of the bulk interception ratios (annual for evergreen conifers and heathlands, seasonal for deciduous woodlands) cited in the literature for each vegetation type. The bulk interception ratios and the resulting γ parameters are given in Table 1. No published estimates of the summer and winter bulk interception in wet woodlands were identified. Therefore, and based on their estimated LAI, bulk interception ratio in summer was assumed to be intermediate between those recorded in beech and oak woodlands during the leafed and leafless seasons, and to be identical to those of beech and oak woodlands in winter. Mixed woodlands were assumed to have LAI, canopy height, mean leaf resistance and bulk interception fraction values intermediate between deciduous and evergreen woodlands.

Following Granier et al. (1999), evergreen needle-leaved species were assumed to have a constant LAI throughout the year, whilst LAI in deciduous broad-leaved species was assumed to increase linearly over 30 days starting at bud break from LAI_{min} to LAI_{max} (Bréda 1994). In autumn, LAI was assumed to decrease linearly over 30 days before leaf fall. Leaf fall was assumed to be complete 10 days after the leaf colouring date. Mean bud break and leaf colouring dates were estimated to be the 10th of April and the 17th of October, respectively, based on figures in Lebourgeois et al. (2010). Based on Janssen (1994, in Spieksma et al. 1996) and informal field observations in the Dauges catchment, grassland LAI was assumed to increase linearly from the start of April to the end of June, and to decrease throughout October. Within the wetland, field observations show that vegetation starts to develop in late May. Full development is only reached in late June-early July. This matches LAI measurements carried out in similar wetlands in western Europe (Nieveen et al. 1998; Sijtsma and Veldhuizen 1992; Spieksma et al. 1997; Bortoluzzi et al. 2006). During transition periods in spring and autumn, crop coefficients, root depths and γ coefficients were linearly interpolated between their minimum (winter) and maximum (summer) values following phenological curves identical to those for LAI. In common with other MIKE SHE-based modelling studies (e.g. Thompson et al. 2004, 2009, 2023; Singh et al. 2010; Chan et al. 2020), all evapotranspiration model parameters were assumed to be constant for each vegetation type from year to year and across the site. Based on data presented by Granier et al. (1999), transpiration from roots was assumed to be increasingly reduced when the soil moisture deficit drops below 0.4.

Model calibration and validation

A detailed account of the original calibration and validation of the model is provided by Duranel (2015) and Duranel et al. (2021), so only a brief summary is provided here. Calibration parameters were selected based on a sensitivity analysis and included Manning's n of the streams represented within MIKE 11, the hydraulic parameters of the saturated zone in both peat and fissured granite (horizontal and vertical saturated hydraulic conductivity, specific yield), and the unsaturated zone parameters for both peat and mineral soils (saturated hydraulic conductivity, volumetric soil moisture at saturation, field capacity and wilting point).

The model was calibrated and validated against observed daily mean discharge and groundwater table depth. Discharge was recorded at the wetland outlet (Pont de Pierre gauging station, Figure 1) and in three small permanent watercourses in the upper part of the catchment (Girrolles, Marzet and Rocher gauging stations, Figure 1). Groundwater table depth below the surface was recorded using automatic loggers in 16 shallow dip wells located within the wetland or on mineral soils on lower slopes. High resolution (15 min) discharge and groundwater table depth

measurements were aggregated to daily means (see Duranel (2015) for more details on hydrological monitoring methods). The model was run from 01/08/1998 (corresponding to the installation of a permanent automatic meteorological station at Saint-Léger-la-Montagne, 4.2 km from the centre of the Duges wetland) to 31/12/2013, including an approximately 2.5-year warm-up period from 01/08/1998 to 31/12/2000. To accommodate slightly different data availabilities, the calibration period was defined as 01/01/2011–30/06/2012 for stream discharge and 01/01/2011–31/12/2012 for groundwater table depth. The validation period was 01/07/2012–31/12/2013 for discharge and 01/01/2013–31/12/2013 for groundwater table depth.

According to model performance guidelines commonly accepted in hydrological modelling literature (Moriassi et al. 2007), model performance with regard to stream discharge was satisfactory (Nash-Sutcliffe efficiency >0.5) to very good ($NSE > 0.75$) depending on the gauging station. In part due to the large number of observation points against which the model was calibrated and validated, performance with respect to groundwater table depth was variable (see Duranel (2015) and Duranel et al. (2021) for a detailed discussion). Here simulation results are considered for seven dip wells located within the mire, where daily groundwater table depth was available for most of the calibration and validation periods, and where model performance was assessed to be satisfactory ($NSE > 0.50$) to very good ($NSE > 0.75$) depending on the period. The locations of these dip wells are shown as D3, D7, D13, D15, D18, D20 and D21 in Figure 1. The root mean square error over the 2011–2013 monitoring period was between 0.068 m and 0.137 m. A sample of observed and simulated discharge and groundwater table depth time-series are shown in Figure 2.

Land use change scenarios

The hydrological impacts of three land use change scenarios were investigated by specifying alternative vegetation distributions within the MIKE SHE / MIKE 11 model. The simulation period employed for each scenario was defined as 01/08/1998 to 31/12/2013 with, as for model calibration / validation, the c.2.5 years up to 31/12/2000 acting as a warm-up period. For each scenario, model results were compared to the baseline, i.e. the calibrated / validated model employing the current, CORINE-based, vegetation distribution. Land use change within the mire was not simulated because for legal, economic and technical reasons large mires in the French Massif Central have rarely been actively afforested. Therefore, the focus of this study is on land use changes within the catchment. The scenarios were defined by replacing all of the current vegetation categories except wetland and wet woodland with conifers, grassland and heathland, respectively. A fourth scenario, corresponding to replacement of existing vegetation except wetland and wet woodland with broadleaf woodland, was also initially investigated. Expectedly, given the current dominance of broadleaf woodland within the catchment, the results of this scenario were broadly similar to the baseline. In the interests of clarity, results from this scenario are therefore not included in the figures and tables below although they are referred to in the text when relevant. The three scenarios that provide the focus of the analysis represent the range of hydrological impacts expected due to plausible changes in catchment land use. They also replicate existing conditions in many mire catchments within the Massif Central, with some catchments having been entirely afforested while others are still dominated by open habitats. Figure 3 shows the distribution of vegetation types under the baseline and three land use change scenarios. The proportions of the entire model area, as well as sub-areas for which water balance analyses were undertaken (see below), covered by each vegetation type for the baseline and land use change scenarios are shown in Table 2.

For each land use scenario, a water balance for the period 2001–2013 was computed using the MIKE SHE water balance tool for two areas upstream of the main wetland outlet located at Pont-de-Pierre in which peat and mineral soils predominate, respectively. The boundaries of these areas are shown in Figure 3. The water balance was not calculated over the entire modelled area for two reasons: first, to reduce potential errors caused by uncertainty in boundary conditions at the north-west end of the modelled area as well as the sparser hydrogeological and hydrological data downstream of Pont-de-Pierre (Duranel 2015); and second, to constrain the focus of the analysis on the majority of designated mire habitats which are upstream of Pont-de-Pierre.

Groundwater table (GWT) depth, exfiltration rate from the saturated zone to overland flow, and upwelling rate from lower to upper computational layers simulated over the 2001–2013 period were mapped across the entire catchment for each scenario and compared to the baseline. Stream flow simulated over the 2001–2013 period at the four gauging stations was summarised using discharge

exceedance curves, discharge quantiles and discharge totals for the baseline and each land use change scenario. A detailed comparison of simulated baseline and scenario GWT depth time series at the locations of seven dip wells focussed on the period 2011–2013 for which, as demonstrated above, the model was determined to satisfactorily reproduce observed conditions (Duranel 2015; Duranel et al. 2021). Scenario impacts on GWT depth were further summarised using exceedance curves for each well that were established using simulation results for the longer 2001–2013 period.

RESULTS

Water balance

Table 3 shows the mean annual simulated water balance computed using the MIKE SHE water balance tool over the period 2001–2013 for both the mire and its catchment (peat and mineral soils upstream of Pont-de-Pierre, respectively) for the baseline and each land use change scenario. Since fluxes are expressed in mm yr^{-1} and the catchment and mire differ in extent, horizontal fluxes (boundary inflow or outflow, flow to or from watercourses) should not be directly compared between catchment and wetland water balances. Figures 4 and 5 show changes in the mean monthly water balance under the different land use scenarios relative to the baseline for the catchment and mire, respectively. Following the MIKE SHE convention, change in storage is positive when increasing, and error is positive when outflows are larger than inflows.

The model simulates substantial impacts on most hydrological fluxes within the mire catchment in response to land use change. Actual evapotranspiration increases by 45% (+249 mm) under the conifer scenario, while it is reduced by 20% (-112 mm) and 17% (-96 mm) under the grassland and heathland scenarios, respectively. The impacts of both the conifer and grassland scenarios are larger in winter, while under the heathland scenario changes are larger in summer. This reflects the much larger difference in interception capacity between conifers and broadleaved trees in winter, the larger differences in interception capacity and crop coefficient between heathland and broadleaved trees in summer, and, for grassland, a larger summer crop coefficient and a greater control of the Penman-Monteith reference evapotranspiration on actual evapotranspiration due to the absence of a distinction between wet-time and dry-time evapotranspiration.

Since within grasslands, and unlike woodland and heathland, interception is not modelled independently of other evapotranspiration fluxes, comparison of individual fluxes within actual evapotranspiration should be made with care. Nevertheless, it is clear that the largest impacts are on interception under the conifer scenario (+96%, +228 mm) and transpiration from the unsaturated zone under the heathland scenario (-20%, -59 mm). The latter differences are due to a lower summer crop coefficient and a smaller root depth for heath compared to broadleaf trees. When compared to the broadleaf scenario, the conifer and heathland scenarios (in which there is no difference in the way actual evapotranspiration is estimated) lead to interception changes of +77% (+203 mm) and -22% (-58 mm), respectively.

Whatever the scenario, evaporation from ponded water and transpiration from the saturated zone are relatively small components of the actual evapotranspiration budget, and therefore inter-scenario differences are very small in absolute terms. Differences in throughfall between land use classes result in differences in surface runoff: the overland boundary outflow (which is principally comprised of surface runoff across the mineral soils/peat soils boundary) is reduced by 50% (-235 mm) under the conifer scenario, while it increases by 29% (+134 mm) and 22% (+102 mm) under the grassland and heathland scenarios, respectively. The impacts of land use changes are most noticeable in winter and spring, following patterns in actual evapotranspiration and, probably, in soil moisture deficit and infiltration capacity. Given the small length of the watercourses that are located on areas underlain by mineral soils, overland flow to watercourses is quantitatively small, but trends under the different scenarios are similar to those described for overland boundary outflow.

Changes in throughfall also impact infiltration from overland to both the unsaturated zone and, when groundwater is at the surface, the saturated zone. The former is reduced by 19% (-180 mm) under the conifer scenario, while it is increased by 21% (201 mm) and 2% (+22 mm) under the grassland and heathland scenarios, respectively. Timing of these changes varies between scenarios, reflecting differences in the seasonal patterns of change in interception. As a result, changes are larger in winter under the conifer scenario, relatively constant throughout the year under the grassland scenario, and larger in summer under the heathland scenario. Changes in percolation

550 from the unsaturated zone to the saturated zone (which, given the limited infiltration from overland
551 water to the saturated zone, can be equated to groundwater recharge) logically follow changes in
552 actual evapotranspiration: it is reduced by 32% (-213 mm) under the conifer scenario, and increased
553 by 14% (+94 mm) and 12% (+80 mm) under the grassland and heathland scenarios, respectively.
554 Seasonal patterns closely replicate those of actual evapotranspiration. Changes in groundwater
555 recharge translate into changes in groundwater outflow, either as saturated zone boundary outflow
556 or as seepage from the saturated zone to the surface. The former is reduced by 13% (-53 mm) under
557 the conifer scenario, while it increases by 3% (+13 mm) and 4% (+15 mm) under the grassland and
558 heathland scenarios, respectively. Saturated zone exchanges to the surface decline by 47% (-
559 157 mm) under the conifer scenario and increase by 23% (+78 mm) and 21% (+69 mm) under the
560 grassland and heathland scenarios, respectively. Seepage from the saturated zone to overland on
561 mineral soil occurs principally in the low-lying areas around the mire boundaries and contributes
562 substantially to overland boundary outflow (Duranel 2015).

563 Hydrological impacts of changes in land use within the catchment translate into impacts on the
564 water balance of the mire. Under the conifer scenario, overland boundary inflows are reduced by
565 half (-50%, -799 mm), while they are increased by 29% (+456 mm) and 22% (+349 mm) under the
566 grassland and heathland scenarios, respectively. In absolute terms, impacts are smaller in summer
567 under the conifer and grassland scenarios, and in spring under the heathland scenario. Groundwater
568 inflows are also impacted: groundwater flow from the lower to the upper computational layer
569 (corresponding to upwelling from the underlying mineral formations to the peat) is reduced by 14%
570 (-149 mm) under the conifer scenario and increased by 4% (+38 mm) and 4% (+46 mm) under the
571 grassland and heathland scenarios, respectively. Fewer MIKE SHE cells are simulated as
572 discharging groundwater under the conifer scenario and there is a 41% (+14 mm) increase in
573 downward flow between computational layers. The opposite occurs under the grassland and
574 heathland scenarios, with a 7% (-3 mm) decrease in downward flow in both cases. In absolute terms,
575 the impact of the grassland and conifer scenarios on groundwater upwelling within the mire is
576 relatively constant throughout the year, albeit with very slightly larger changes in autumn. This
577 reflects, with some temporal buffering, the timing of changes in groundwater recharge throughout
578 the catchment. The smaller upwelling inputs to the peat layer under the conifer scenario result in
579 reduced seepage from the saturated zone to overland flow (-17%, -163 mm), whereas this seepage
580 increases under both the grassland (+4%, +42 mm) and heathland (+5%, +53 mm) scenarios.

581 Even though none of the scenarios include change in land use within the mire, they still result
582 in small changes in actual evapotranspiration from the mire: it is reduced by 2% (-13 mm) under the
583 conifer scenario and increased by +2 mm (+0%) and +4 mm (+1%) under the grassland and
584 heathland scenarios, respectively. In all cases the impacts are larger in summer, and effectively
585 negligible in winter. As discussed below, this is a consequence of changes in summer GWT depths
586 within the mire, which themselves result from changes in the inputs to the mire described above.
587 Interception within the mire, as expected, does not change between scenarios since it is only driven
588 by precipitation and canopy characteristics, which are identical across the mire in all scenarios.
589 Evaporation from ponded water is reduced under the conifer scenario (-12%, -52 mm) and increased
590 under the grassland (+2%, +8 mm) and heathland (+5%, +20 mm) scenarios, suggesting that the
591 mire surface is less and more frequently flooded, respectively. Transpiration from the unsaturated
592 zone is increased (+29%, +28 mm) under the conifer scenario while it is reduced under the grassland
593 (-4%, -4 mm) and heathland (-11%, -11 mm) scenarios. This points to changes in the proportion of
594 the mire where the GWT depth is below the ground surface such that an unsaturated zone can
595 develop and transpiration from this zone can occur. The simulated changes in transpiration from the
596 saturated zone (increased under the conifer scenario and reduced under the grassland and
597 heathland scenarios) may seem initially counter-intuitive given the latter explanation. However, this
598 is a consequence of the order in which individual evapotranspiration fluxes are used to fulfil potential
599 evapotranspiration, and of the proportion of potential evapotranspiration that is still available for
600 transpiration from the saturated zone once all other components have been satisfied. Changes in
601 GWT depth and, consequently, the depth and extent of the unsaturated zone within the mire can
602 also explain changes in infiltration from the surface to the unsaturated zone and in percolation from
603 the unsaturated zone to the saturated zone. Both these fluxes increase under the conifer scenario
604 (+27%, +43 mm and +24%, 15 mm, respectively) but reduce under the grassland (-4%, -6 mm and -
605 4%, -2 mm, respectively) and heathland (-9%, -14 mm and -5%, -2 mm, respectively) scenarios.

Changes in catchment land use also have consequences on outflows from the mire. The model simulates a 27% (-677 mm) reduction of overland to stream outflows under the conifer scenario, whereas these outflows increase by 13% (+321 mm) under the grassland scenario and 10% (+260 mm) under the heathland scenario. Almost all water balance error is caused by the overland flow component, and is the result of lack of convergence in some model cells due to the use of the successive over-relaxation method to decrease model run times (see Duranel et al. 2021). This error is small on mineral soils (1.6% of total inputs for the baseline), giving high confidence in simulated inputs to the mire, but is larger within the mire (9.9% of total inputs) due to the shallower slopes. In all scenarios, the water balance error is negative, meaning that total outputs (and more specifically total outputs from the overland flow component) are underestimated. The absolute overland error within the mire is reduced under the conifer scenario (due to reduced inputs to the overland flow component) and increased under the grassland and heathland scenarios (for the opposite reason). Overland flow outputs (in particular overland flow to watercourses) are therefore underestimated to a lesser degree under the conifer scenario compared to the baseline, and under the baseline compared to the grassland or heathland scenarios. Consequently, simulated differences between scenarios in terms of overland outflow to watercourses should be viewed as a minimum. This is important when considering differences in simulated stream discharge, since it means that the impacts of changes in catchment land use simulated by the model are, if anything, slightly underestimated, particularly for the wetland outlet at Pont-de-Pierre.

Simulated stream discharge

The model simulates substantial impacts of changes in catchment land use on stream discharge (Figure 6, Table 4). Replacement of the current catchment vegetation by conifer plantations would result in a reduction in discharge at all four gauging stations. Mean discharge is reduced by between 28% and 47% depending on the station. Impacts are larger in absolute terms for high flows compared to low flows, but larger in percentage terms for low flows with declines in Q95 (i.e. discharge that is equalled or exceeded 95% of the time) ranging between 46% and 82% compared to an equivalent range of declines for Q5 (discharge equalled or exceeded for 5% of the time) of 22-45%.

The grassland and heathland scenarios result in increased mean, Q5 and Q95 discharges at all gauging stations although the magnitude of changes is smaller than for the conifer scenario. Mean discharge increases by between 13% and 29% under the grassland scenario and by between 9% and 26% under the heathland scenario. Again, the impact is generally larger in absolute terms for high flows but larger in percentage terms for low flow, although this pattern is less clear than under the conifer scenario. Increases in high flow (Q5) are larger under the grassland scenario than under the heathland scenarios whereas increases in low flows (Q95) are larger for the heathland scenario. For example, Q95 increases by between 13% and 31% under the grassland scenario but by between 33% and 89% under the heathland scenario. This is due to the larger interception capacity and therefore increased actual evapotranspiration for heathland in winter compared to grassland, and conversely a larger root depth and summer crop coefficient and therefore increased actual evapotranspiration for grassland in summer compared to heathland (Table 1, Figure 4).

In percentage terms, the impact of the different scenarios on discharge is larger at the upstream gauging stations (Girolles, Rocher and Marzet) than at the catchment outlet (Pont-de-Pierre). However, this may be explained, at least in part, by the previously discussed smaller absolute overland flow error within the mire under the conifer scenario than under the baseline, and under the baseline than under both the grassland and heathland scenarios.

Simulated groundwater table depth

Figures 7, 8 and 9 show the spatial distribution of change compared to the baseline in simulated mean monthly GWT depth (2001–2013) across the whole model domain under the different land use change scenarios. The same results are shown in Figures 10, 11 and 12 but with a geographical focus on the peat soils upstream of Pont-de-Pierre. Overall, the conifer scenario results in a drop in GWT depths within both the wider catchment and the mire (Figures 7 and 10). The grassland (Figures 8 and 11) and heathland (Figures 9 and 12) scenarios result in higher GWT levels, with the exception of areas where grassland under the baseline is replaced by heathland and vice-versa.

The largest overall impacts of changes in land use on GWT depth are around the catchment boundaries, and smallest in topographic depressions, especially within the main mire. Within the mire, the largest impacts are around the wetland boundaries, particularly in small peripheral extensions of the peat away from the centre of the mire. Changes in GWT are smaller, but still occur, in the centre of the mire and at its downstream end. On mineral soils within the wider catchment, the impacts of the conifer scenario are largest in late autumn and winter (maximum mean anomaly in December), and smallest in summer and early autumn (minimum mean anomaly in September). A similar pattern occurs under the grassland scenario, with the largest impacts in November and the smallest in August. Under the heathland scenario, changes in GWT depth are largest in autumn (maximum mean anomaly in October) and smallest in late winter and early spring (minimum mean anomaly in April). Within the wetland seasonality of change is different: in all cases impacts are largest in late summer and early autumn. The largest changes occur in September for the conifer and heathland scenarios, and in October for the grassland scenario. The smallest changes occur in January for the conifer and grassland scenarios and in April for the heathland scenario.

Figure 13 shows simulated GWT depths at the location of selected dip wells within the mire for the baseline and each land use change scenario. These data are shown for the period 01/01/2011–31/12/2013 (i.e. the combined calibration and validation period for which, as described above, model performance in terms of GWT at these locations is at least satisfactory – see also Duranel et al. 2021). As shown in Figure 1, dip wells D3 and D13 are located relatively close to the mire boundary, D7, D15, D18 and D20 are further towards the mire centre, while dip wells D18 and D21 have an intermediate position. Figure 14 shows the simulated GWT depth exceedance frequency curves for the same dip well locations, but for the entire simulation period (excluding the warm-up period – i.e. 2001–2013). The percentage of time during which simulated GWT depth at each dip well location falls below specified depths below ground level is shown and for each scenario in Table 5.

Model results for the seven dip wells further demonstrate that land use change within the catchment impacts GWT depths within the mire. These impacts are strongly seasonal, are largely restricted to the 3–4 driest months of the year, generally in summer and early autumn, and vary with location. Impacts are large in dip wells located in sloping parts of the mire and close to its margins, particularly at D3, but also D13 and to a lesser extent D21. Changes are much smaller (D7, D15, D18) or almost non-existent (D20) at the locations of dip wells that are further into the centre of the mire. As revealed by the previous review of changes in monthly mean GWT, the largest impacts are simulated under the conifer scenario, with a drop in GWT depth close to the mire margins. For example, the model suggests that in the autumn of 2011, coinciding with a period of drought, GWT at D3 drops by up to an additional 1.3 m compared to the baseline (Figure 13). In the wetter years of 2012 and 2013, GWT depth at D3 (as well as D13, also close to the mire's margin) drops less markedly, by up to an additional 0.6 m. In winter, when the peat is saturated to the surface, land use change within the catchment has no or very little effect. At this time of year, inputs to the mire far exceed outputs through evapotranspiration and groundwater losses to the river network (Duranel 2015), and very large reductions in inputs from surface runoff and groundwater upwelling would be required to have a noticeable impact on GWT depth within the peat. Across the selected dip wells, and compared to the baseline, the amount of time when GWT is below the ground surface and then further than 0.1 m and 0.2 m below ground level increases by 7 to 36 days per year (increases from the baseline of between 4% and 41%), 3 to 48 days (3–124%) and 3 to 49 days (4–178%), respectively (Table 5).

Compared to the conifer scenario, the impacts of the grassland and heathland scenarios on GWT depths within the mire are much smaller. It should be noted that the scope for change in GWT depth in the peat soils in response to the increases in inflows to the mire reported above for these two scenarios is relatively limited since the GWT is already at, or very close to, ground level for most of the year under baseline conditions. As a result, additional inputs to the mire for much of the time will not increase GWT but instead contribute to the enhanced groundwater exfiltration and saturation-excess surface runoff to watercourses revealed by the water balance analysis. Nevertheless, both scenarios generally result in a slight reduction compared to the baseline in the magnitude of the seasonal fall in groundwater levels during the driest months of the year. For instance, under the baseline and during the 2011 drought year, the GWT at dip well D3 drops to about 30 cm below ground in autumn, whereas it is maintained at ground level under the heathland scenario (Figure 13). At dip wells D18 and D20, the grassland scenario actually results in a very slight increase in the number of days (4, 3 and 4, respectively) when the GWT is deeper than 0.0, 0.1 and 0.2 m below

ground (Table 5). Given that these increases are only simulated for these two dip wells, and that the water balance analysis indicates a general increase in inputs to the mire under the grassland scenario, the reason for this is likely linked to the water balance error associated with the overland flow component of the model. At the locations of all other dip wells, this scenario leads to a reduction in the number of days when the GWT exceeds these thresholds (up to 11, 13 and 12 days). The heathland scenario also results in a reduction of these durations for all dip wells of up to 19 days.

DISCUSSION

Parameterisation of the evapotranspiration model

In the absence of an independent calibration of the evapotranspiration model against observed interception, transpiration or soil moisture content, it is important that model parameters are carefully chosen based on the best published evidence in order to reduce uncertainties. In the current study, the HYLUC γ parameter was calibrated against the mean of bulk interception ratios (annual for evergreen conifers and heathlands, seasonal for deciduous woodlands) cited in the literature for each vegetation type (Table 5 in Supplementary Materials). The simulated interception fractions can therefore be considered as representative of average conditions in similar habitats, but may differ from those within the Dauges catchment. The literature suggests remarkably similar annual bulk interception ratios in Douglas fir plantations in a range of Western European countries (0.37 ± 0.02), possibly because of relatively standardised forestry practices for this species. Therefore, the confidence associated with this value for conifers within the Dauges catchment is relatively high. Reported interception ratios in broadleaf woodlands are more variable (0.28 ± 0.07 in summer, 0.16 ± 0.08 in winter), probably due to varied forestry practices and stand characteristics (Bansept 2013). The applicability of an average interception ratio for broadleaf trees within the Dauges catchment is therefore more uncertain. Conversely, the absence of data to parameterise interception in wet woodlands, and the assumptions that had to be made as a result, are likely to have negligible consequences on model results given that the area occupied by this vegetation class is small and does not vary between land use scenarios.

Estimation of the β parameter (or basal crop coefficient) was more complex. As its name suggests, the crop coefficient method used in both HYLUC and MIKE SHE to estimate actual evapotranspiration in the absence of water stress was primarily designed for agricultural applications (Allen et al. 1998). In contrast to widely cultivated crops, for which a large volume of data is available, crop coefficients have rarely been estimated for natural and semi-natural vegetation, including forests. In the current study, these coefficients were estimated based on climate, vegetation height, leaf area index and mean leaf resistance using equations proposed by Allen et al. (1998) and Allen and Pereira (2009). This method suggested almost identical basal crop coefficients for deciduous and coniferous woodlands in summer (0.77 and 0.78, respectively), which agreed with the value of 0.75 used by Granier et al. (1999) for both broadleaved and coniferous species in BILJOU, the reference forest ecohydrology model used in France. Based on a literature review and their own data, Granier et al. (1999) showed that the ratio of transpiration to reference evapotranspiration (in their case Penman's reference evapotranspiration assuming an albedo of 0.11) in the absence of water stress is relatively constant for stands with LAI equal to or exceeding 6, and similar in broadleaved and coniferous species. Kantor (1995, cited by Katzensteiner et al. 2011) came to similar conclusions for mature Norway spruce and beech stands. Other researchers have also found T/ET_{ref} ratios in the absence of water stress of around 0.7-0.8 in both broadleaved and coniferous trees (Aussenac and Boulangeat 1980; Čermák et al. 1982; Nizinski et al. 1989; Bréda et al. 1993; Biron 1994; Bréda and Granier 1996; Ladefkarl 1998; Haria and Price 2000).

Very little data were found to assess the reliability of basal crop coefficient estimates obtained for forests in winter (Table 1 in Supplementary Materials). Tiktak and Bouten (1994) suggested slightly lower values in winter than in summer for Douglas fir. For broadleaved woodlands, values range from 0 (Nizinski et al. 1989) to 0.82 (Calder et al. 2002). The 0.38 value used in this study is compatible with some residual transpiration from the understorey and tree trunks (Kozłowski 1943) and increased evaporation from ponded water during the leafless season. Given the relatively low reference evapotranspiration in winter, the uncertainty associated with this value is unlikely to have a large impact on model results. Similarly, values available in the literature to validate basal crop coefficients in wet woodlands are very variable. However, as previously noted, wet woodlands only

cover a small proportion of the Dauges catchment and this does not vary within the different land use scenarios. Consequently, uncertainty associated with crop coefficients for this vegetation class is also unlikely to have major impacts on model results.

Basal crop coefficients estimated for heathland were 0.60 and 0.69 in winter and summer, respectively. These values are compatible with T/ET_{ref} ratios measured or estimated in similar habitats across Western Europe (Table 1 in Supplementary Materials). For instance, Calder (1986) measured annual values in a Scottish heather-rich moorland (based on Penman grass reference evapotranspiration) that ranged between 0.58 and 0.67 whilst Jansen (1994, in Spieksma et al. 1996) measured a T/ET_{ref} ratio of 0.6 in a dry heathland in the Netherlands (based on Penman open water reference evapotranspiration). Researchers who have investigated differences in ET/ET_{ref} or T/ET_{ref} ratios also found only slightly lower values in winter than in summer (Eggink and Vink 1989; Janssen 1994, in Spieksma et al. 1996; Calder et al. 2002). Grassland crop coefficients (0.6 in winter and 0.9 in summer) were based on values measured by Janssen (1994, in Spieksma et al. 1996) in oligotrophic acidic grassland on sandy soils in the absence of water stress. These values are intermediate between the reference values recommended by the FAO (Allen et al. 1998) for extensively grazed pastures (0.3 and 0.75 in winter and summer respectively) and for rye-grass hay meadows (0.95 and 1.05), and within the range of values measured by Pôças et al. (2013) in mountain grasslands in Portugal.

The forest scenarios used in the current study assume complete coverage of catchments by woodland that is mature enough to form a closed canopy. In reality, even in catchments where forestry is the dominant land use, open habitats are generally present due to clear-cutting, particularly in conifer plantations (e.g. Swift et al. 1975; McCaughey 1981). Model results should therefore be viewed as representing the likely maximum hydrological impact of land use change at the catchment scale in general and on mire hydrological conditions in particular.

Simulated impacts of catchment land use on water yield

The simulated impacts of the different land use scenarios on water fluxes within the catchment are consistent with a number of observational studies worldwide. For instance, simulations show that 44%, 64%, 45%, 37% and 35% of annual precipitation is lost through evapotranspiration under the baseline and the conifer, broadleaf, heathland and grassland scenarios, respectively. The values for conifers and grassland are almost identical to those reported by Harsch et al. (2009) from a 40-year long lysimeter experiment in Germany: 65% in a pine forest and 36% in a grassland. The result for the broadleaf scenario (45%) is, however, smaller than that observed in this earlier study in an oak-beech forest (56%). The ratio of annual transpiration over precipitation simulated by the model for the broadleaf scenario (23%) is, however, compatible with those observed in European beech stands under similar annual precipitation and reviewed by Schipka et al. (2005). Ratios ranging from 17% to 25% were reported in areas with annual precipitation in the order of 1200–1300 mm.

In the current study, simulated groundwater recharge is 53%, 36%, 52%, 59% and 60% of annual precipitation under the baseline and the conifer, broadleaf, heathland and grassland scenarios, respectively. The difference between the conifer and broadleaf scenarios is slightly larger than that recorded by Kantor (1995, cited by Katzensteiner et al. 2011) in mature Norway spruce and beech stands in the Czech Republic (16% vs. 10%). It corresponds to a 205 mm yr⁻¹ difference in absolute terms and is also substantially larger than the 60–100 mm yr⁻¹ difference obtained by Rosenqvist et al. (2010) when comparing Norway spruce and common oak stands in Denmark and Sweden. However, the soil and climate conditions in these previous studies differed markedly from those within the Dauges catchment. Cooper (1980) recorded groundwater recharge that was 44% lower under a Scots pine plantation than under a nearby grassed clearing on sandy soils in eastern England, a figure very similar to the 41% decrease in simulated groundwater recharge for the conifer scenario when compared to the grassland scenario in the Dauges catchment.

Simulated water yield (here defined as the sum of overland boundary outflow and stream discharge from the mineral catchment) for the grassland scenario is 387 mm yr⁻¹ and 148 mm yr⁻¹ larger than those of the conifer and broadleaf scenarios, respectively. In a meta-analysis of 145 deforestation experiments worldwide, Sahin and Hall (1996) reported an average increase in water yield of 330 mm yr⁻¹ after removal of conifer forests and of 201 mm yr⁻¹ and 169 mm yr⁻¹ after removal of broadleaf forests in areas where mean annual precipitation was below and above 1500 mm, respectively. However, across all of the studies reviewed, a wide range of changes was reported

with this range being particularly large for broadleaf compared to conifer forests. In a review of 26 afforestation experiments carried out in paired catchments worldwide, Farley et al. (2005) found that annual runoff was reduced by on average 44% ($\pm 3\%$) when grasslands were afforested. The Dauges model simulates relatively similar values: when compared to the grassland scenario and depending on the gauging station, simulated mean stream discharge is reduced by between 13% and 25% under the broadleaf scenario and by between 36% and 57% under the conifer scenario. Reductions in discharge are larger in the small upstream reaches.

Low flows are most affected in relative terms: the model simulates a decrease in Q95 of between 12% and 30% under the broadleaf scenario and between 52% and 85% under the conifer scenario, with the largest impacts, once again, being simulated in the upstream reaches. These results are in accord with the findings of a literature review of the effects of forestry on river low flows undertaken by Johnson (1998). This concluded that most clear-felling experiments produced increases in low flows with the magnitude of changes varying between 15% and 350% depending on the area of the catchment felled and the catchment's physical characteristics. Robinson and Dupeyrat (2005) identified increases in Q95 of 20% and 40% for two Plynlimon catchments (Wales) following felling of 26% and 49% of total catchment area (approximately half the forested area in each case given the original 48% and 99% coverage of conifer plantations).

Overall, the simulated impacts of the different land use scenarios on water fluxes within the Dauges catchment are compatible with the observational evidence available from similar environments. Although a few studies suggest a larger impact of the broadleaf forests that dominate current land use on water fluxes than those simulated, this is not unexpected given the large variation in forestry practices and stand characteristics in these forests, as discussed above. The impacts simulated as part of this study should, therefore, be viewed as representative of average forestry practices. Notwithstanding these issues, a key concern highlighted by the results is the potential for an expansion in tree cover, especially coniferous forests, to reduce downstream river flows, especially in summer when discharge is already low (i.e. the large simulated declines in Q95). Thompson et al. (2026) have shown that declining summer stream flows dominate the climate change signal for the Dauges catchment. There is, therefore, potential for larger declines than those reported herein if coniferous plantations were combined with projected changes in climate. Such additive impacts of multiple perturbations could be explored using the MIKE SHE / MIKE 11 model.

Although the Dauges catchment is relatively small, it is broadly representative of many upland catchments and their mires across the granitic Massif Central that sustain larger river systems. A number of communities in the region have had water supplies, especially in summer, repeatedly impacted by drought (Meunier 2001; TF1 2023) whilst further afield, warnings have been issued that the exacerbation of low flow conditions could impact cooling of nuclear power stations, including the Civaux plant on the Vienne River of which the Dauges catchment is a headwater (Cour des Comptes 2023). The potential for exacerbating these water resources problems will clearly need to be considered, alongside the implication for upland peatlands that are discussed below, in the formulation of land use and forestry policies.

Impacts of catchment land use on peatland hydrological conditions

Despite the growing body of literature demonstrating the differential impacts of forests and open habitats on the hydrological cycle, only a handful of studies worldwide have investigated the impact of catchment afforestation on hydrological or ecological conditions within wetlands, and direct evidence is patchy and weak.

Smith and Charman (1988) showed that for blanket bogs surrounded by conifer plantations in Northumberland (UK), the proportion of species they classified as "ombrogenous" (but are in fact characteristic of relatively wet conditions in acidic mires generally speaking) was inversely proportional to the length of time elapsed since conifers were planted. Conversely, the proportion of species characteristic of drier mire habitats was larger in mires surrounded by older conifer plantations. Using the MMS/PRMS hydrological model, Helmschrot (2006) evaluated potential impacts of catchment afforestation in a primarily grass-covered South-African catchment on several hydrogeomorphic wetland types. Runoff, interflow and groundwater discharge to wetlands were predicted to decline in nearly all cases. Krause et al. (2007) modelled several complex land use scenarios, including increases or reductions in forest cover, on the hydrology of a large floodplain wetland in Germany. They found only minor effects on the floodplain water balance as alterations in

vertical groundwater flow were counter-balanced by both lateral groundwater flow and the close interactions between surface water and groundwater within the floodplain. In contrast, Grenfell et al. (2006) used a simple rainfall-runoff model to show that surface and subsurface inflows to a small hillslope seepage wetland in South Africa may have been considerably reduced following the plantation of *Eucalyptus* trees across the entire catchment. Discrepancies between wetland boundaries mapped using floristic and edaphic indicators also indicated that the extent of wetland plant communities may have been reduced following afforestation. Conversely, in an observational study in Patagonia, Weigandt et al. (2015) failed to identify reductions in surface runoff to hillslope wetlands surrounded by pine plantations compared to those in which grassland dominated the catchment. The impacts of catchment land cover change upon the hydrology, and in turn, ecology of wetlands is therefore likely to be conditioned by a range of factors. These include, most obviously, the nature of the changes (e.g. physical extent of the land cover changes, location of changes with respect to the wetlands, vegetation types involved) but also the hydrogeological setting and baseline hydrological conditions including the water balance and water level regime.

The current study provides much needed evidence on the potential impacts of land cover change on peatlands within headwater catchments of basement regions, environments which are underrepresented in the literature. It demonstrates that catchment afforestation can have substantial impacts on the water balance by reducing surface and groundwater inflows to the hillslope, depression or valley bottom wetlands that are often found in these regions (Pinault 2025). It shows that plantations of conifer species with a large interception capacity may have a more detrimental impact than semi-natural broadleaf woodlands. Changes in GWT depth within the wetland due to catchment land use change are particularly noticeable in late summer, a time when the GWT is already relatively deep, and when recent climate change impact assessments suggest a dominant trend towards lower GWT depths (Thompson et al. 2026). Again, the simulation of combined land use and climate change scenarios would be a useful future extension of research employing the Dages MIKE SHE / MIKE 11 model.

Given the sensitivity of mire vegetation to GWT, including seasonal draw-downs (e.g. Bragazza and Gerodi 1999; Hilbert et al. 2000; Hose et al. 2014), the declines simulated following afforestation may have important consequences for biodiversity conservation should water tables become “out of range” with respect to requirements of specific wetland plant species or communities (e.g. Wheeler et al. 2004; Clilverd et al. 2022; Thompson et al. 2023). Results from the current study therefore offer a potential explanation to Smith and Charman’s (1988) findings related to the decline in wetland species over time following afforestation. Interestingly, these authors also showed that species indicative of wetter conditions were more frequent in mires with a relatively short boundary with the forest compared to their area. This supports the findings of the current study whereby larger impacts of land use changes on GWT were simulated along the mire margins. Ongoing research within the Dages catchment seeks to establish the hydrological (e.g. water level) controls upon specific peatland vegetation species. This would facilitate an assessment of spatially distributed, species-specific, impacts of land cover change via the combination of the mire-wide simulated hydrological conditions for each scenario and peatland vegetation hydrological requirements using the approaches described by Clilverd et al. (2022) and Thompson et al. (2023).

Model results suggest that the impacts of land cover change on stream discharge are larger upstream of the mire compared to downstream of it whilst the hydrological changes that modify recharge of the catchment’s fissured granite aquifer have a larger impact on GWT depth close to the mire margins compared to at its centre. Duranel et al. (2021) demonstrated that under baseline conditions these marginal areas experienced some of the greatest upwelling of groundwater and are exemplars of edge-focussed discharge (Richardson et al. 2001) or margin seepage (Hare et al. 2017). Such focussed groundwater influxes can be critical in maintaining water table levels and surface saturation in upland wetlands (Devito et al. 1996). This suggests that any monitoring or surveillance schemes aimed at detecting environmental changes resulting from perturbations within the mire’s catchment, for instance land use changes such as those investigated herein but also groundwater abstraction, surface water diversions, or climate change, should include mire margins rather than just the central areas of the peatland, as well as small headwater tributaries above the mire. Long-term surveillance of GWT depth in observation wells located within the fissured zone upstream of the mire could also provide an early warning system for hydrological changes within the catchment, even before their impacts begin to be manifest within the mire.

Changes in GWT depth within the mire due to land use change within the catchment may have effects on the superficial peat geochemistry with the potential for further knock-on impacts. For example, declines in GWT depths in acidic mires have been linked to increased carbon exports to the atmosphere and to downstream watercourses (Freeman et al. 2001; Clark et al. 2005, 2009; Strack et al. 2008; Fenner et al. 2013; Nieminen et al. 2017), and to releases of heavy metals, metalloids and radionuclides sequestered in the saturated peat (Owen and Otton 1995; Rothwell et al. 2009, 2010, 2011; Schöner et al. 2009). In this way, widespread land use changes across the region, in particular coniferous afforestation as revealed by this study, could enhance radiative forcing as well as contributing to additional water resources management problems. The implications of enhanced element release to surface waters could, for example, be amplified by reduced dilution because of the declines in discharge discussed above (Whitehead et al. 2009).

The current study clearly demonstrates that policies aimed at the conservation of mire habitats in sites of similar hydrogeomorphic settings to the Dauges mire, and more largely in groundwater- and surface water-fed wetlands, should encompass their whole catchment. For example, Article 6 of the EU 92/43/EEC Habitats Directive states that “any plan or project [...] likely to have a significant effect thereon, either individually or in combination with other plans or projects, shall be subject to appropriate assessment of its implications for the site in view of the site's conservation objectives”. The present study shows that any activity likely to result in changes in groundwater flow within granite weathering formations upstream of protected mires, even beyond SAC boundaries, falls within the remit of this article. This includes changes in land cover within the catchments of these mires, in particular the conversion of open habitats to woodland or the conversion of broadleaf forest to coniferous plantations. Policies should be implemented to avoid long-term cumulative impacts of small-scale forestry operations that may, if considered individually, not have a significant impact or fall below area thresholds set by environmental legislation or regulators. Afforestation programmes for upland areas are growing in number and extent as a means of addressing climate and other environmental targets (e.g. Valette et al. 2024; Glinel 2025; Syder et al. 2025). Results from the Dauges catchment demonstrates that such programmes may impact the hydrology of upland peatlands with consequences for the ecosystem services and conservation benefits these wetlands provide. There is therefore potential for conflicts between different environmental and conservation policies supporting the argument that robust, scientifically grounded, trade-off mechanism will be required to balance competing demands and priorities (e.g. Iversen et al. 2024; Minter et al. 2025).

CONCLUSIONS

Hydrological modelling using the MIKE SHE / MIKE 11 system has shown that changes in catchment land use may have significant impacts on the water balance and hydrological conditions of groundwater- and surface water-fed peat-based wetlands within an upland catchment of the Massif Central. Results suggest that planting conifer species with a high interception capacity may have a particularly detrimental impact. This could include a reduction in both overland and groundwater inflows to the mire, leading to a lowering of mire groundwater tables, especially during the already relatively dry summer period. Some of the largest changes are concentrated around the wetland margins. These findings have implications for the development and implementation of policies and practical measures aimed at protecting wetland sites of high conservation value. They demonstrate the need to consider interactions between wetlands and their wider catchments rather than just focussing on the geographical limits of conservation sites. This imperative to take a catchment-wide approach is strengthened given the potential for climate change to further impact the hydrology of upland catchments and their mires. Further research should be undertaken to investigate any ecological and geochemical changes that may already have taken place in sites impacted by catchment afforestation, or the impacts that could occur in the future with expansion of conifer plantations, in order to inform assessments of potential ecosystem impacts driven by the hydrological changes revealed by the current study.

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 1724

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1726
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1732
1733 **Competing Interests:** The authors have no relevant financial or non-financial interests to disclose.
1734

1735 **Author Contributions:** All authors contributed to the study conception and design. The original
1736 MIKE SHE / MIKE 11 model was developed by Arnaud Duranel under the supervision of Julian
1737 Thompson and Helene Burningham. Arnaud Duranel led field data collection and survey work and
1738 the installation of monitoring equipment with assistance from Philippe Durepaire and Hervé
1739 Cubizolle. Arnaud Duranel undertook the land cover change scenario simulations and preliminary
1740 analysis. Arnaud Duranel and Julian Thompson prepared the first draft of the manuscript which was
1741 then edited and approved by all of the other authors.

1742
1743 **Data Availability:** The datasets generated during and/or analysed during the current study are
1744 available from the corresponding author on reasonable request.

Figure captions

Figure 1. The Dauges catchment and hydrological monitoring equipment.

Figure 2. Observed and simulated discharge and groundwater table depth at a sample of monitoring locations within the Dauges catchment. The vertical line separates the calibration and validation periods.

Figure 3. Distribution of vegetation types within the modelled area under current conditions (baseline) and three land use change scenarios.

Figure 4. Simulated impacts of changes in catchment land use on the catchment mean monthly water balance (2001–2013, absolute changes relative to baseline). OL: overland; SZ: saturated zone; UZ: unsaturated zone.

Figure 5. Simulated impacts of changes in catchment land use on the mire mean monthly water balance (2001–2013, absolute changes relative to baseline). OL: overland; SZ: saturated zone; UZ: unsaturated zone.

Figure 6. Simulated flow exceedance frequency curves (2001–2013) at the four stream gauging stations for the different catchment land use scenarios.

Figure 7. Spatial distribution of change relative to the baseline in simulated monthly mean groundwater table depth (2001–2013) under the conifer scenario.

Figure 8. Spatial distribution of change relative to the baseline in simulated monthly mean groundwater table depth (2001–2013) under the grassland scenario.

Figure 9. Spatial distribution of change relative to the baseline in simulated monthly mean groundwater table depth (2001–2013) under the heathland scenario.

Figure 10. Spatial distribution of change relative to the baseline in simulated monthly mean groundwater table depth (2001–2013) within the mire under the conifer scenario. Only grid cells corresponding to peat soils upstream of Pont-de-Pierre are shown. Grid cells on mineral soils have been masked out. The colour scale has been skewed to aid visual clarity and interpretation.

Figure 11. Spatial distribution of change relative to the baseline in simulated monthly mean groundwater table depth (2001–2013) within the mire under the grassland scenario. Only grid cells corresponding to peat soils upstream of Pont-de-Pierre are shown. Grid cells on mineral soils have been masked out. The colour scale has been skewed to aid visual clarity and interpretation.

Figure 12. Spatial distribution of change relative to the baseline in simulated monthly mean groundwater table depth (2001–2013) within the mire under the heathland scenario. Only grid cells corresponding to peat soils upstream of Pont-de-Pierre are shown. Grid cells on mineral soils have been masked out. The colour scale has been skewed to aid visual clarity and interpretation.

Figure 13. Simulated groundwater table depth (2011–2013) in selected dip wells within the mire for the different catchment land use scenarios.

Figure 14. Simulated groundwater table depth exceedance frequency curves (2001–2013) for selected dip wells within the mire for the different catchment land use scenarios.

Figures

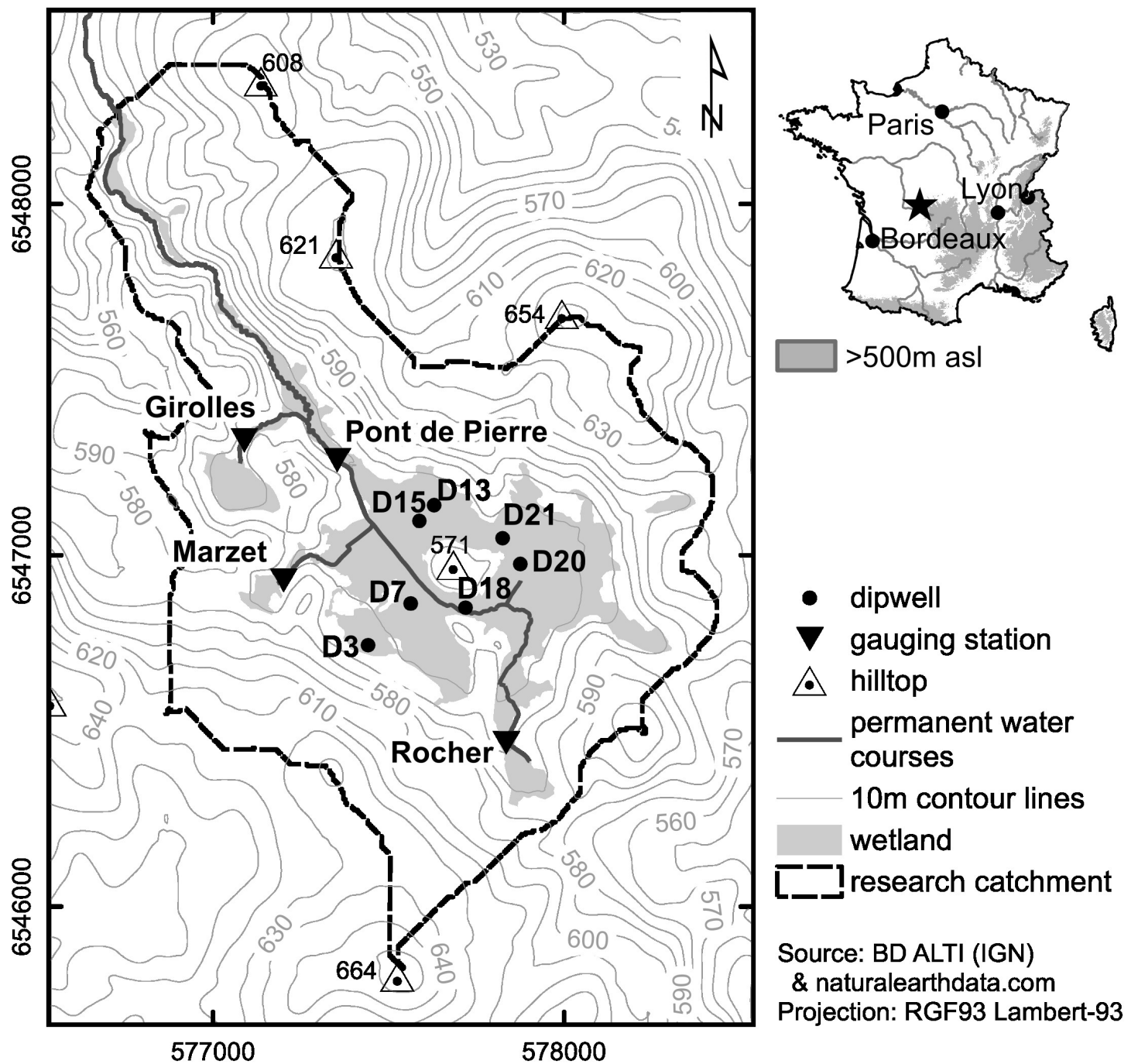


Figure 1

The Dauges catchment and hydrological monitoring equipment.

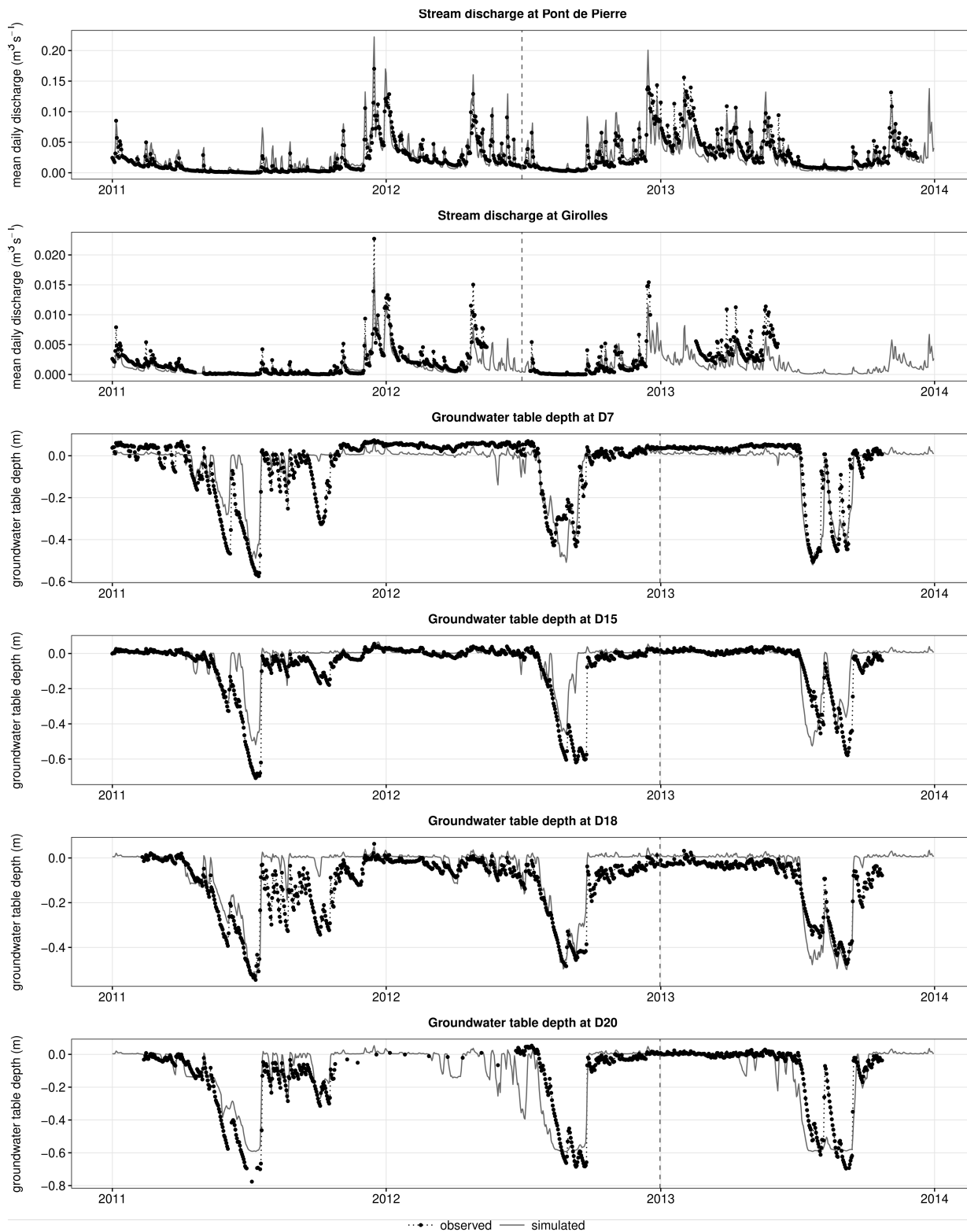


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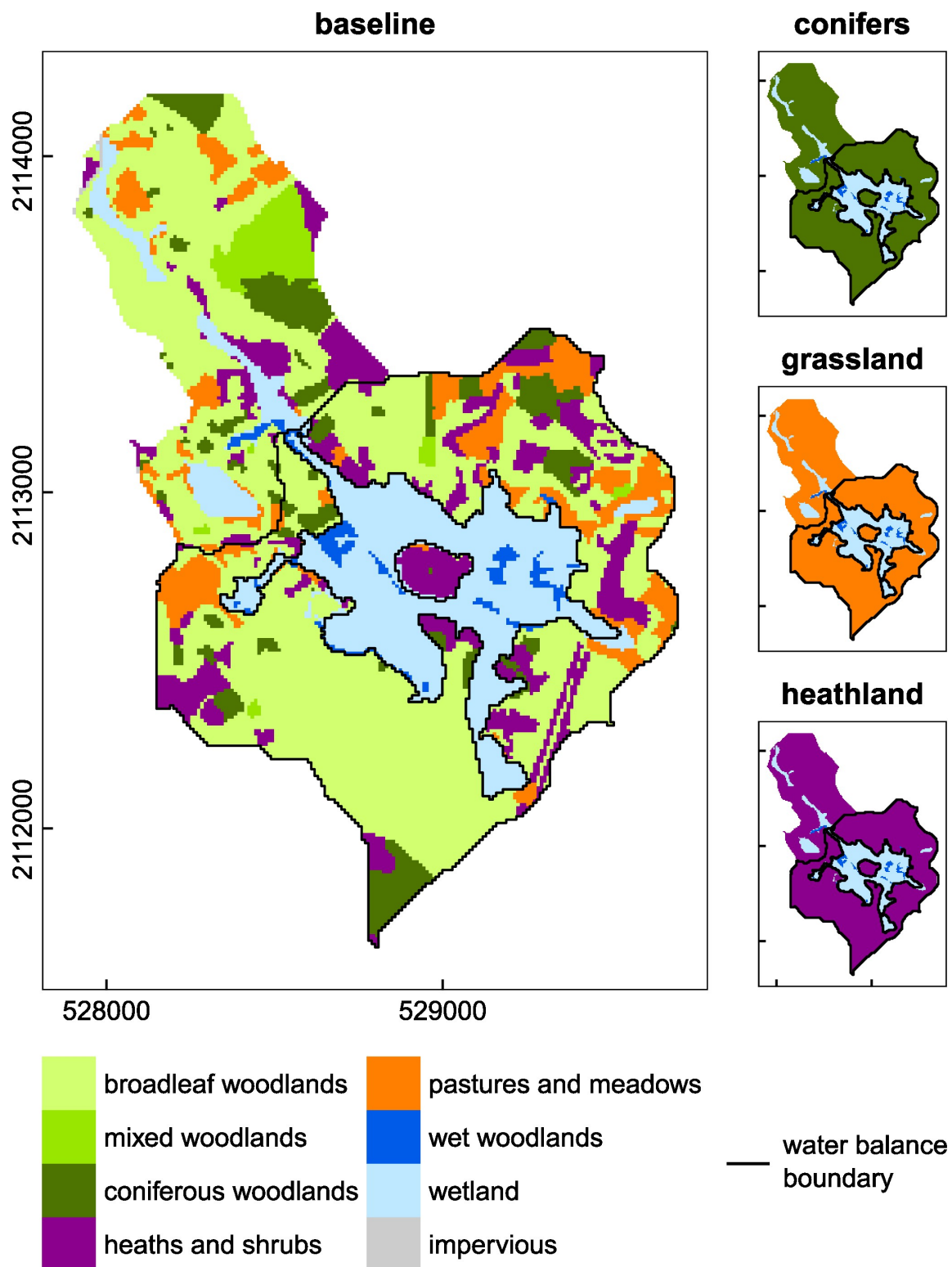


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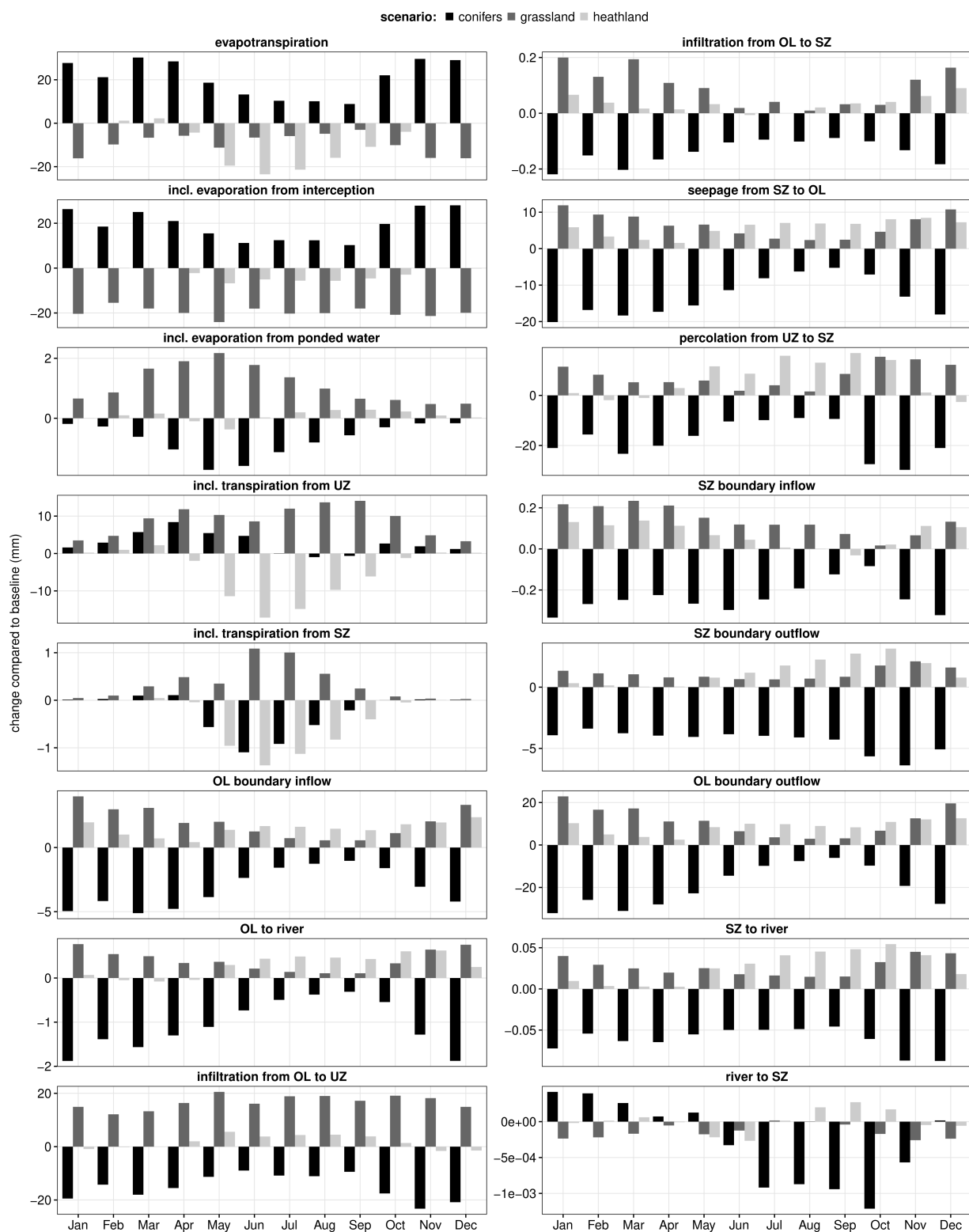


Figure 4

Simulated impacts of changes in catchment land use on the catchment mean monthly water balance (2001–2013, absolute changes relative to baseline). OL: overland; SZ: saturated zone; UZ: unsaturated zone.

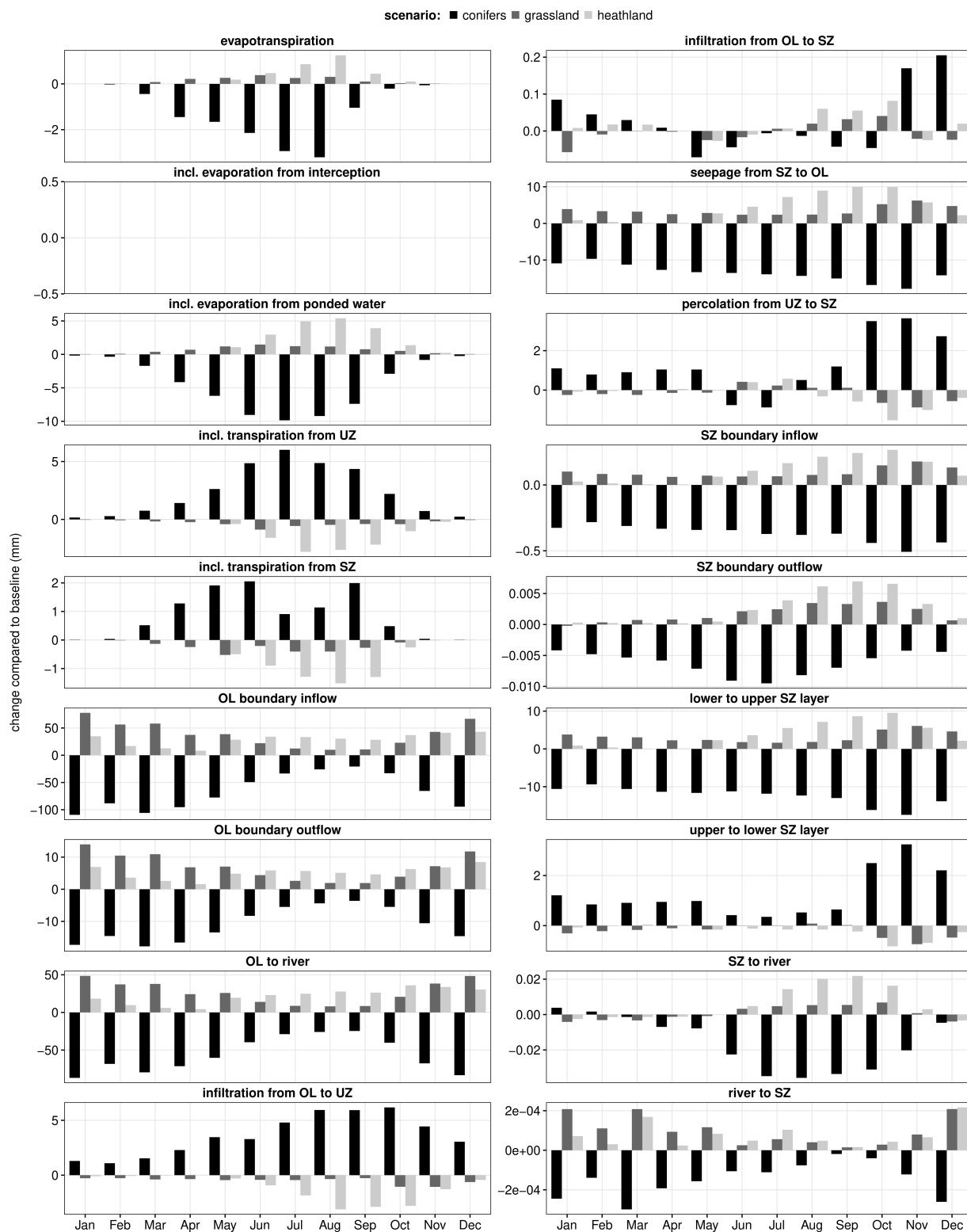


Figure 5

Simulated impacts of changes in catchment land use on the mire mean monthly water balance (2001–2013, absolute changes relative to baseline). OL: overland; SZ: saturated zone; UZ: unsaturated zone.

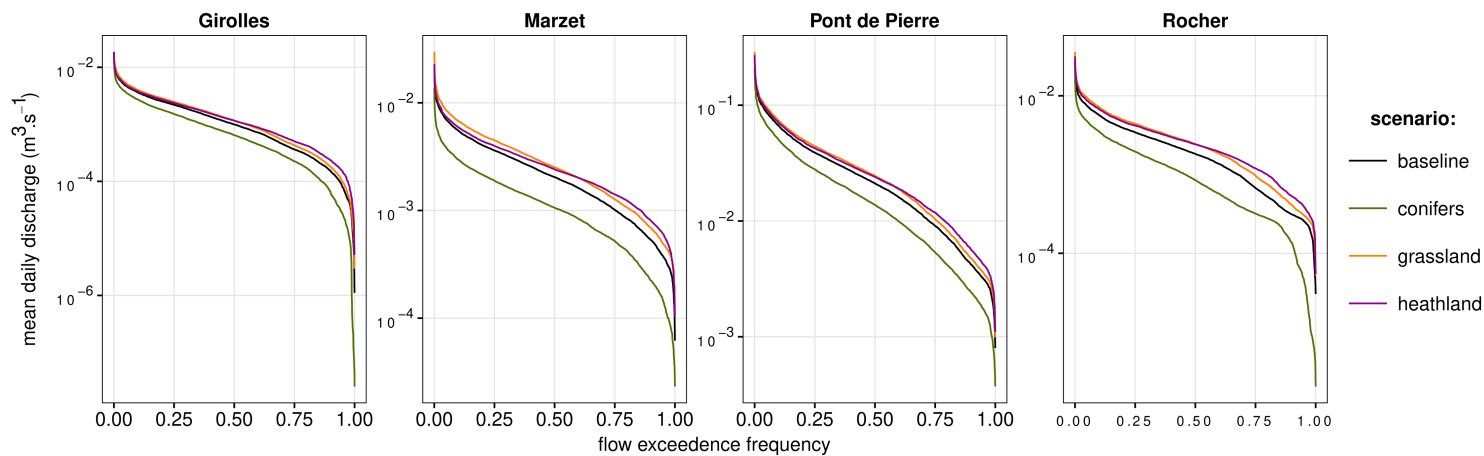


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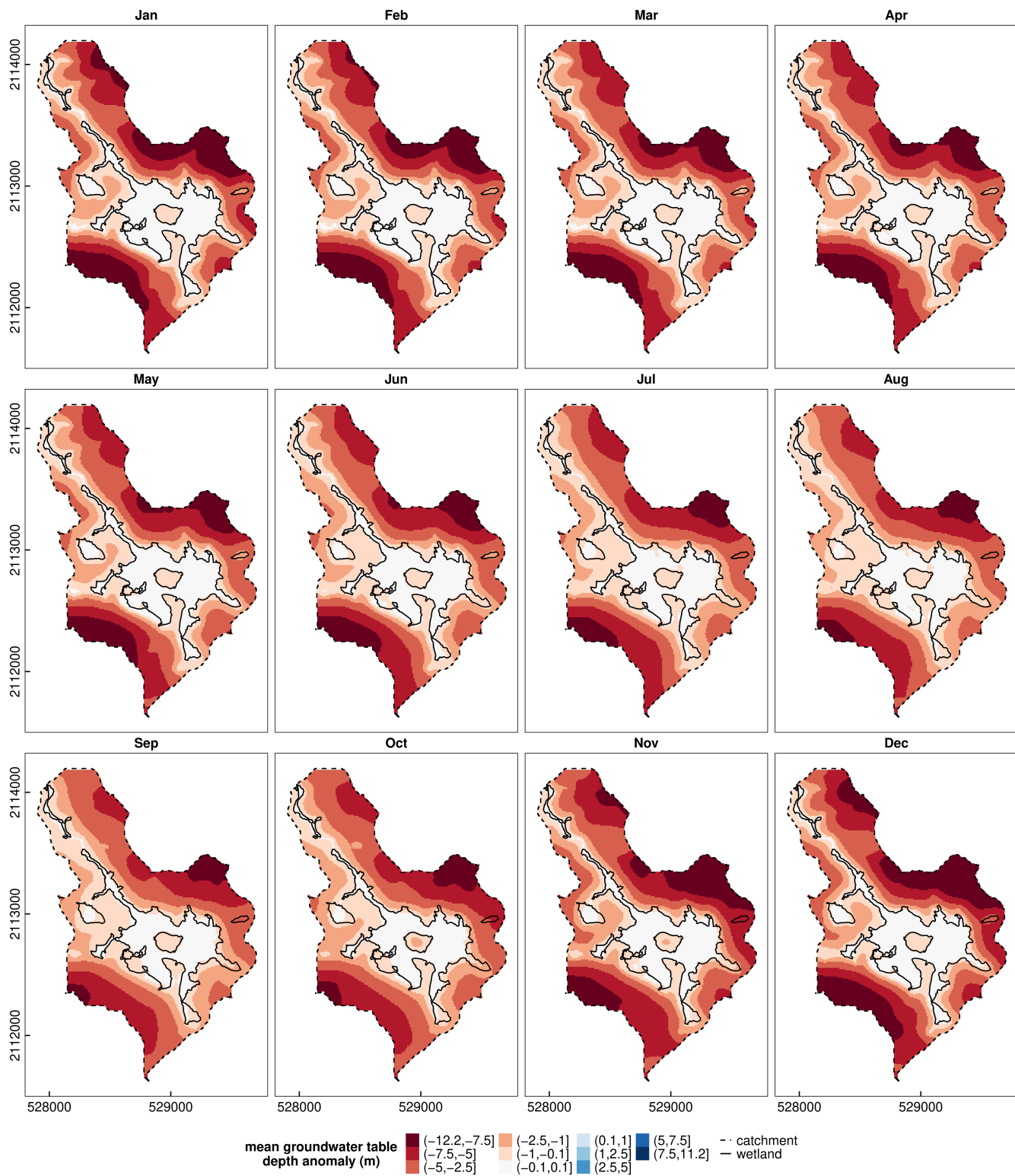


Figure 7

Spatial distribution of change relative to the baseline in simulated monthly mean groundwater table depth (2001–2013) under the conifer scenario.

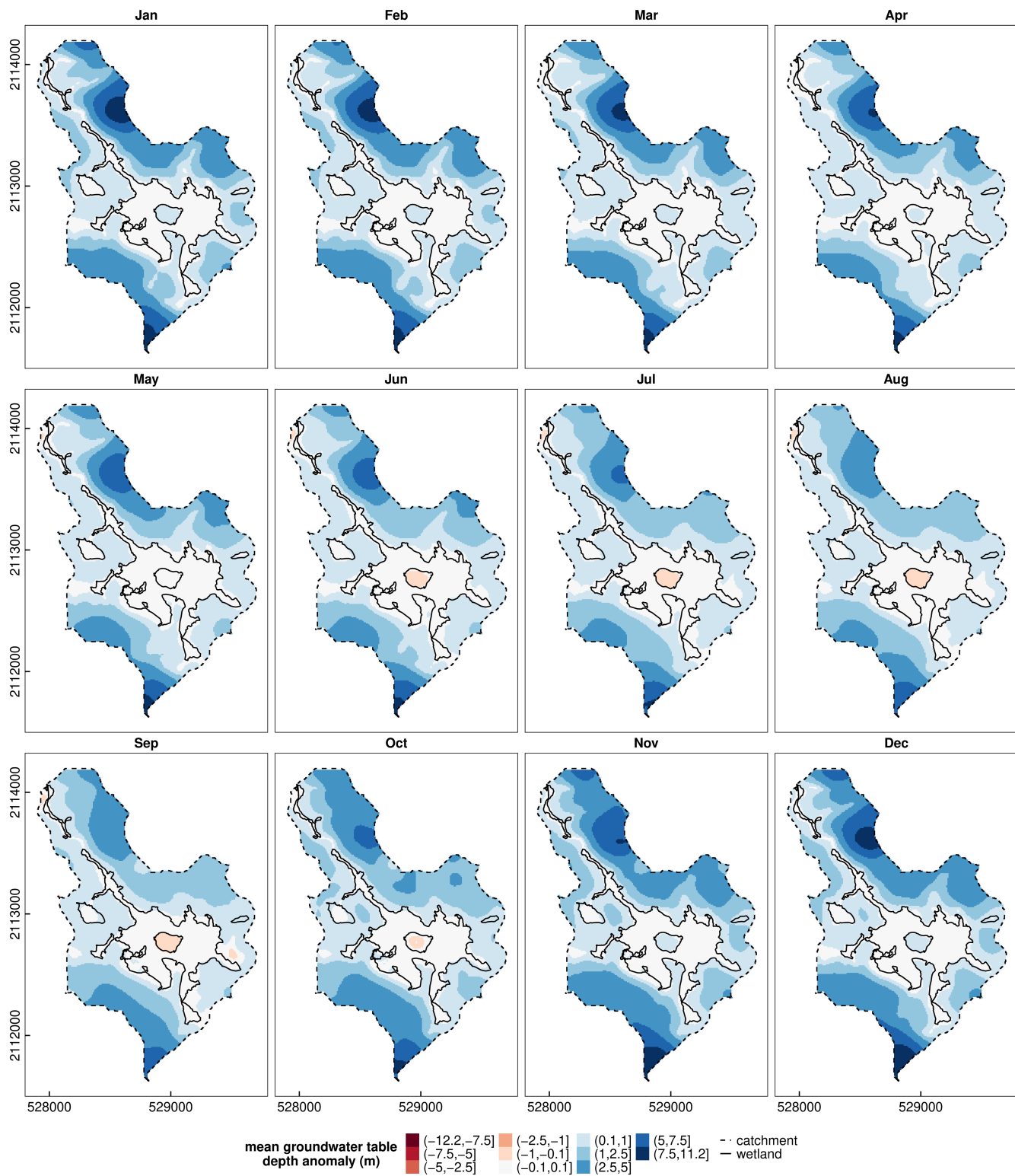


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Spatial distribution of change relative to the baseline in simulated monthly mean groundwater table depth (2001–2013) under the grassland scenario.

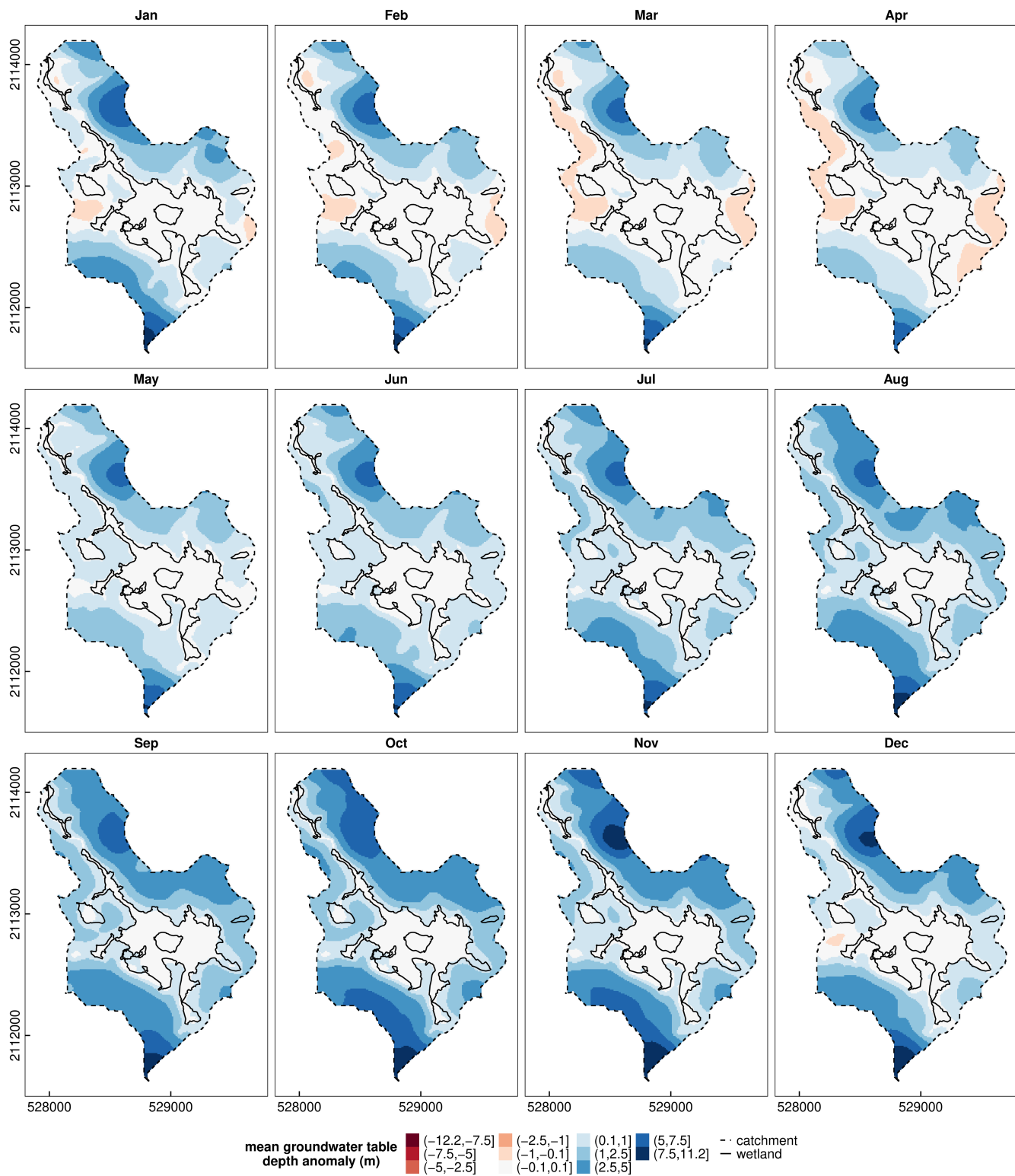


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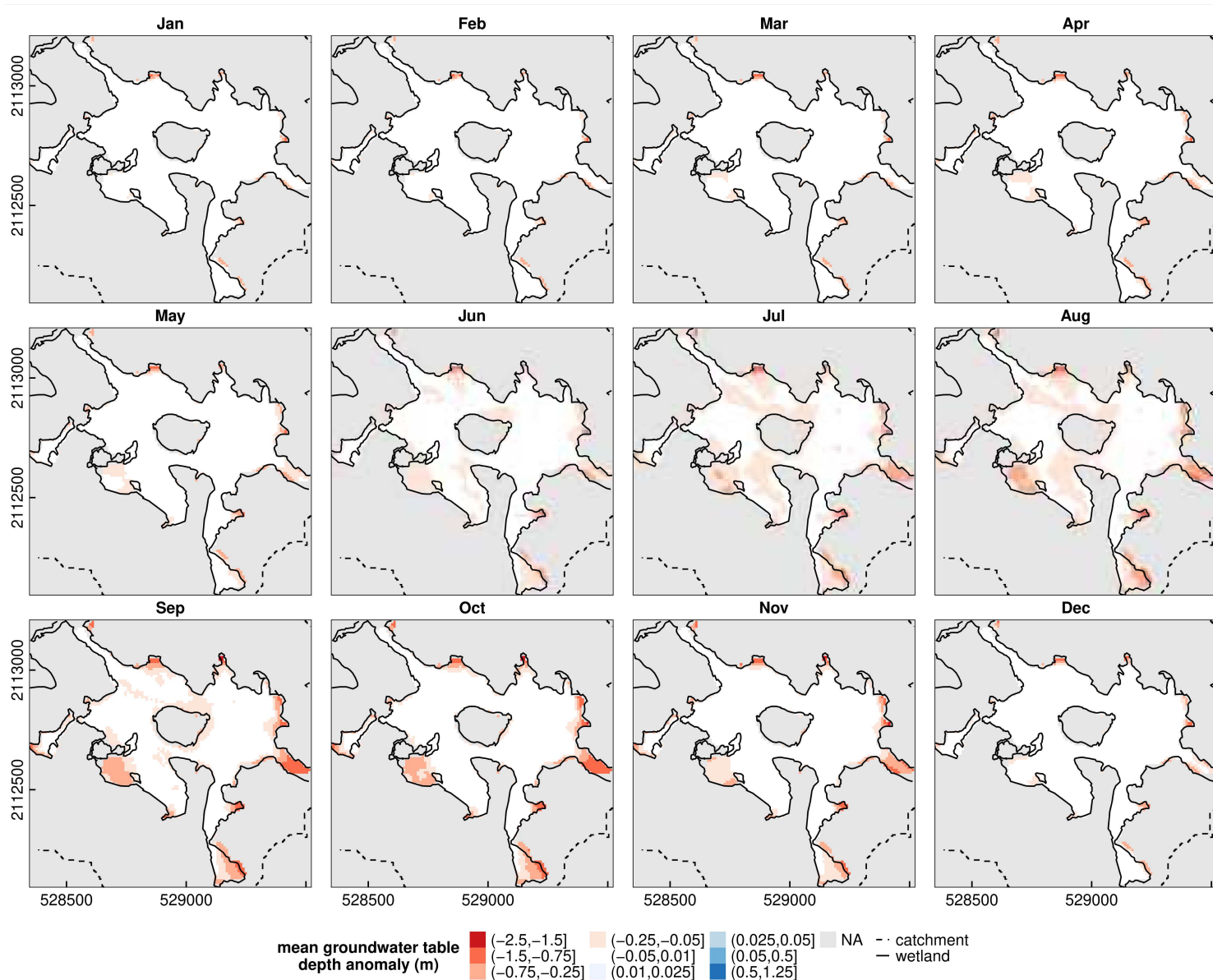


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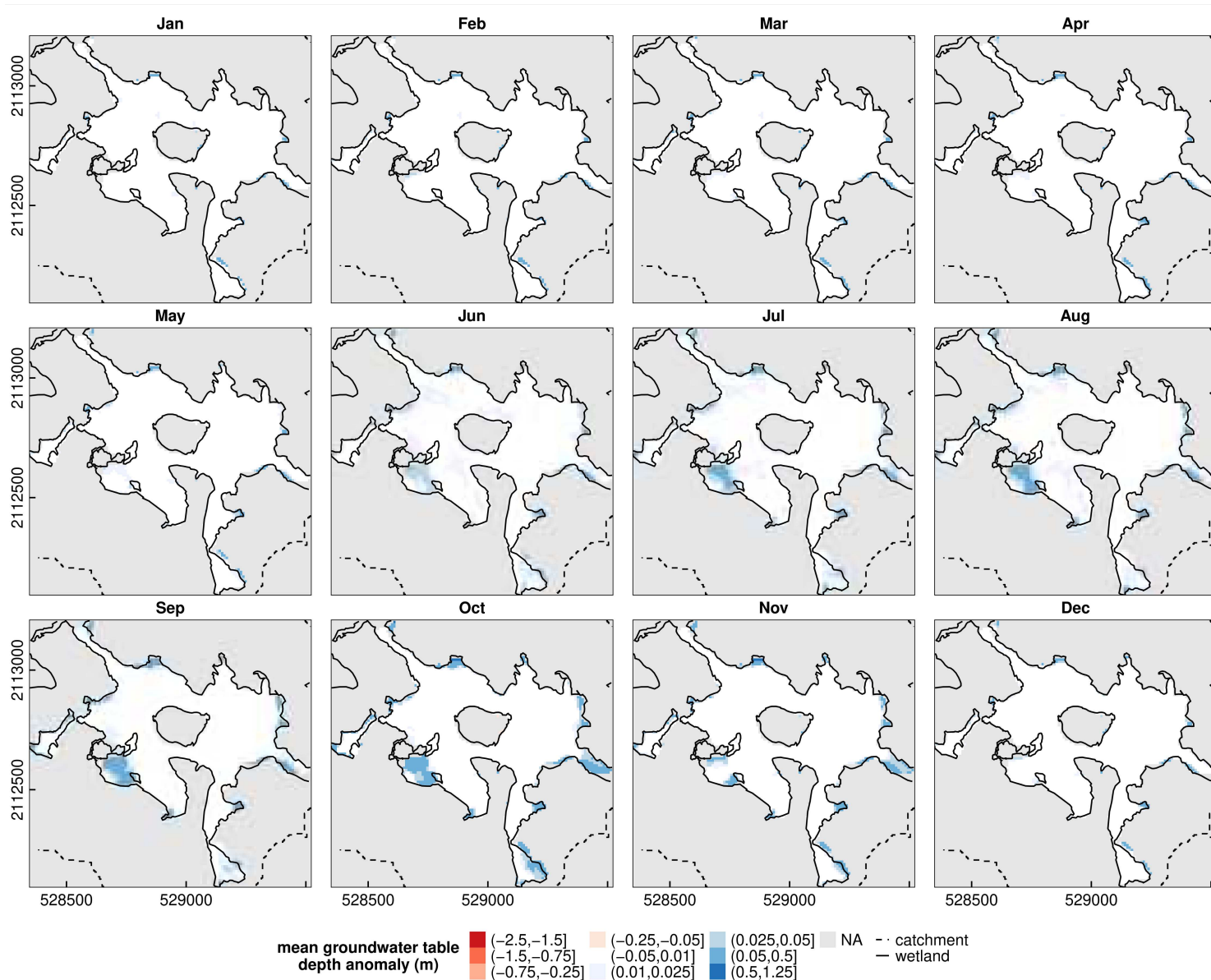


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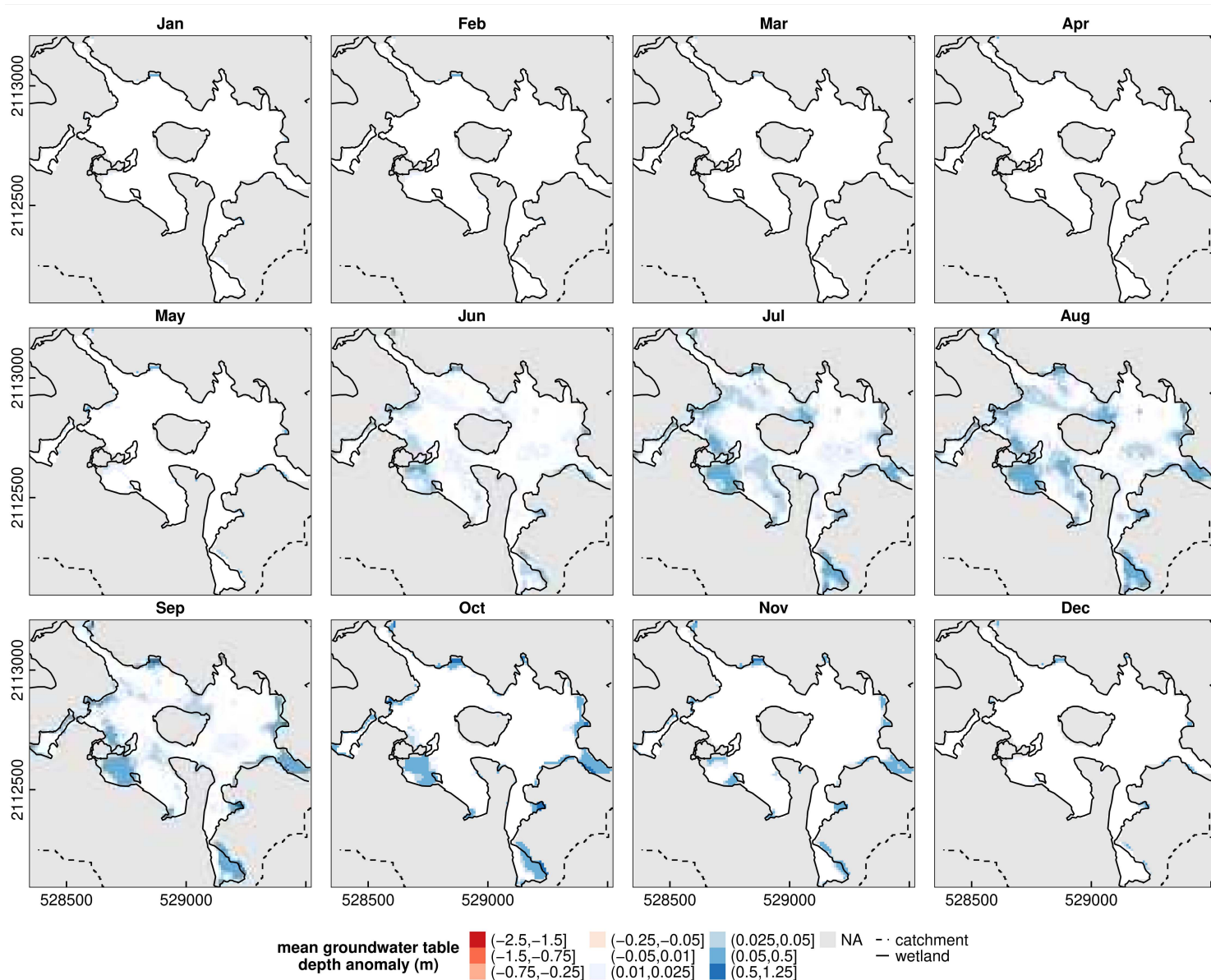


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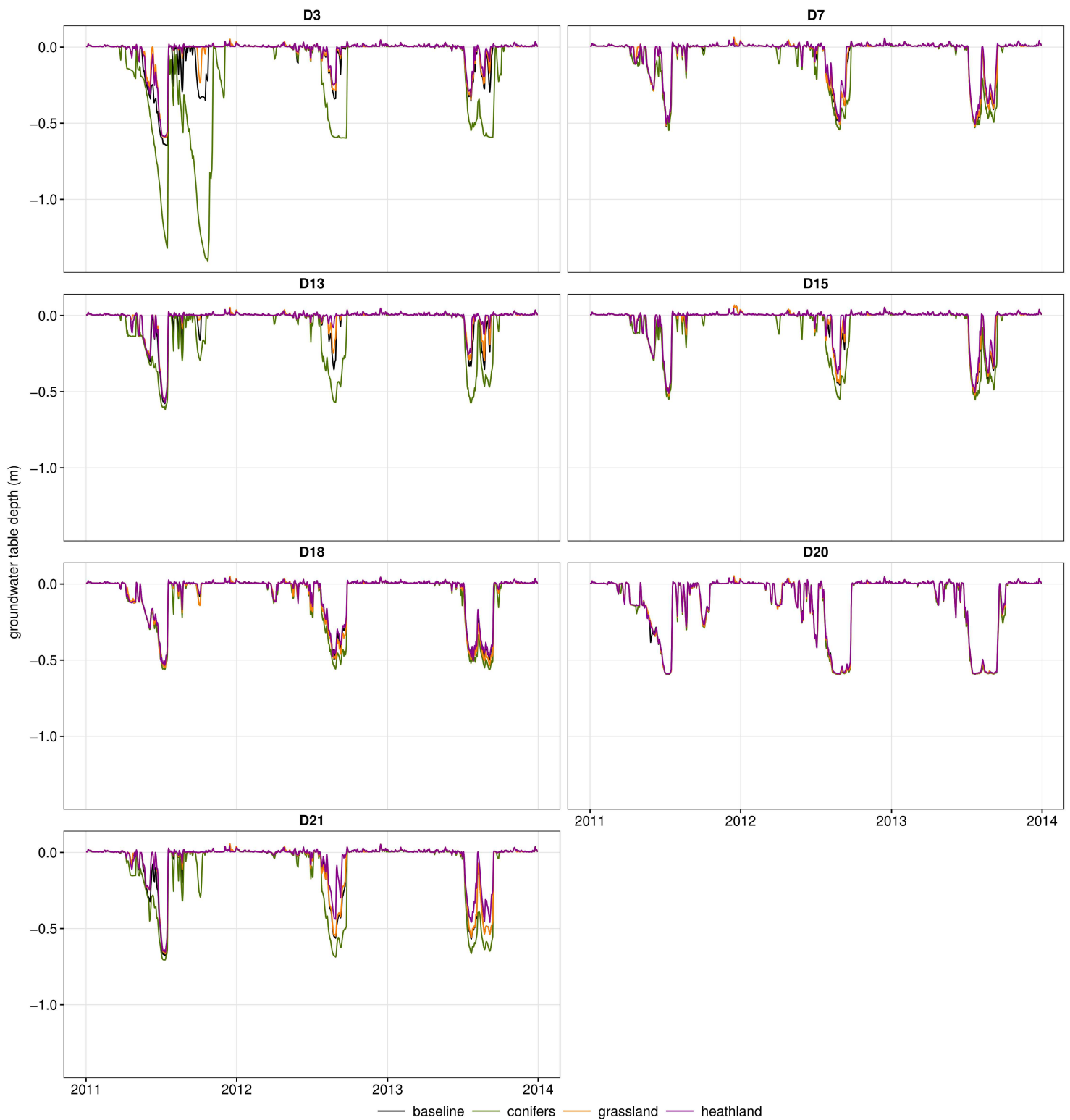


Figure 13

Simulated groundwater table depth (2011–2013) in selected dip wells within the mire for the different catchment land use scenarios.

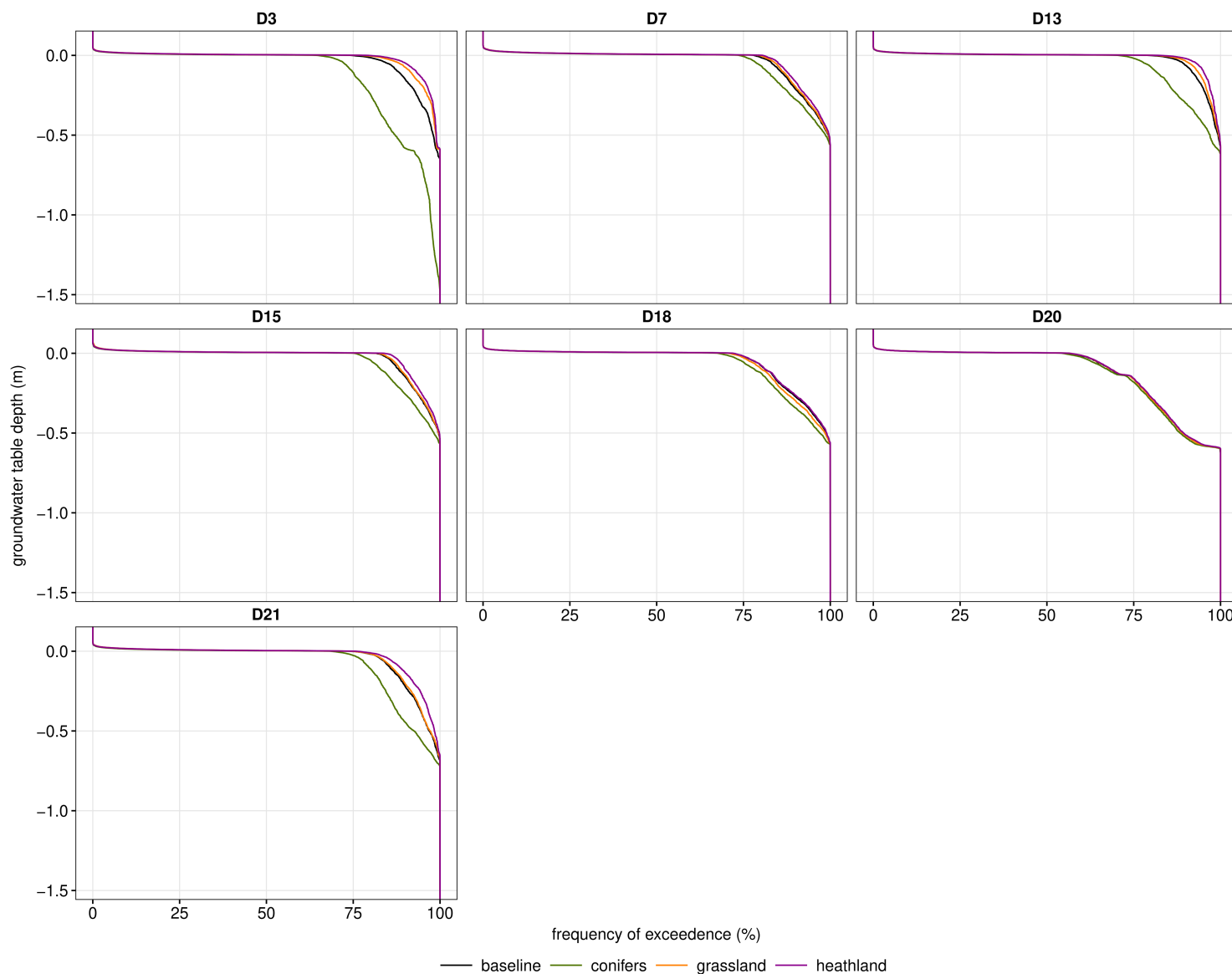


Figure 14

Simulated groundwater table depth exceedance frequency curves (2001–2013) for selected dip wells within the mire for the different catchment land use scenarios.

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