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Review of spatial prioritisation methods for Ecosystem-based Adaptation measures to drought risks

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Abstract

This review focusses on landscape-scale spatial prioritisation methods (SPM's) for Ecosystem-based Adaptation (EbA) measures to drought risks. SPM's are methods that allow to identify priority areas for the conservation or restoration of certain functions on a landscape scale. In PROWATER, priority areas are defined as those areas where a measure has the greatest benefit in terms of reducing drought risk per unit surface area. Each type of adaptation measure has its own prerequisites in terms of abiotic characteristics (e.g. topography, soil) that make the area suitable for the targeted hydrological function. Especially in intensively used areas where available space is limited it is important to take efficient measures, or when funds are scarce (Walter et al. 2007; Wilson et al. 2011).

1. Introduction

Many landscapes in Western Europe have been altered for agricultural intensification and urban development. These changes have decreased the resilience of hydrological systems and have had enormous impacts on biodiversity (Figure 1). Land use changes and land use intensification often coincide with increased soil sealing (urbanisation), soil compaction, increased interception (plantations), groundwater abstraction and drainage of wetlands (Figure 1). But most importantly, these changes lead to a decrease in the natural water availability. Runoff and drainage lead to declining (ground)water levels and the degradation and loss of wetlands. This degradation has in its turn impact on river hydrology, soil nutrient retention, soil carbon sequestration and biodiversity. Loss of biodiversity is not only associated with the physical loss of habitats, but also with the degradation of its quality (desiccation, eutrophication and acidification) and additional greenhouse gas emissions.

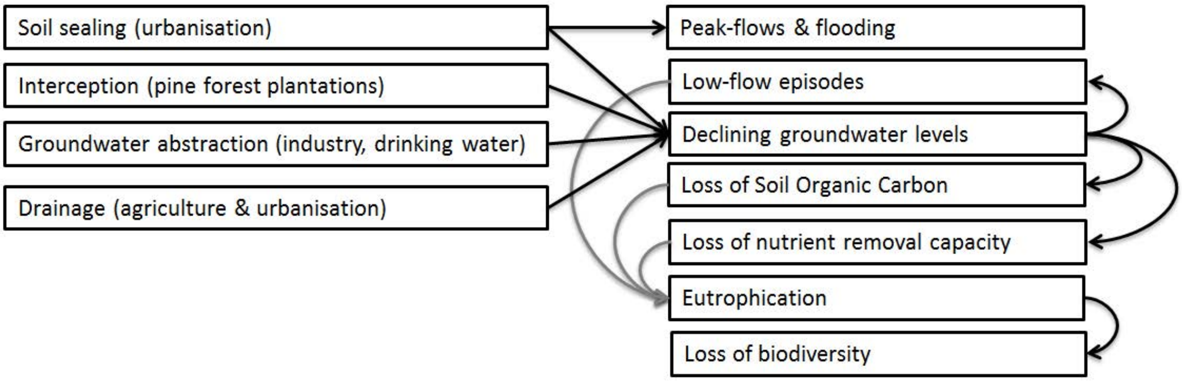


Figure 1: Pressures on the water system and related impacts

When strategic water reservoirs and/or aquifers are sufficiently replenished, drought periods and associated water demands can be bridged. But the replenishment of these strategic water reserves (lakes and/or aquifers) has become insufficient because our landscapes have been degraded and are not adapted to deal with extreme weather.

To increase water security, we need to make better use of occasional periods with extreme precipitation surplus to offset the shortages in the summer water balance. Storage and treatment of water in artificial reservoirs will not suffice. We need to decrease run-off and drainage to store more water in the landscapes. This will enhance natural purification and aquifer recharge (figure 2).

Ecosystem-based land use planning and management is considered as an important measure to increase resilience against flooding, droughts and associated water quality problems. The physical system thus needs to be considered as a structuring factor in land use planning. Several concepts have been established in respect to land use patterns and functionality such as Land Quality (Bouma 2002; Bouma 2006) and Leakiness Indices (Dumanski and Pieri 2000; Doran 2002). Shared by these approaches is that fluxes of water and substances determine the sustainability at certain locations given a certain land use. Land use patterns that reckon with the physical properties of soil and hydrology cause less interaction with the water system whilst a high discrepancy between actual land use and physical suitability urges a more intense adaptation of the system and thus to a higher impact of land use on the water system. A smart implementation of measures can thus provide an important element for climate adaptation strategies.

The challenge for management and planning is to restore a natural diversity of ecosystems and create (semi-artificial) opportunities for ecosystem development that can compensate for climate changes and anthropogenic impact. This Ecosystem-based Adaptation (EbA) approach thus requires a new approach for land use planning that includes spatial objectives for the multitude of services that need to be generated on the limited land surface.

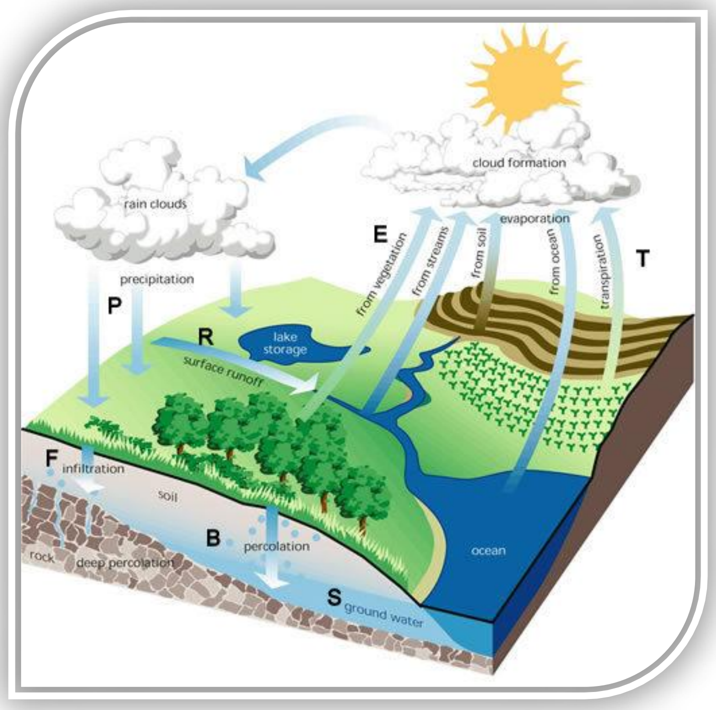


Figure 2: schematic representation of the various hydrological processes that drive the global water cycle.

PROWATER aims to provide leverage to the implementation of measures that increase retention and infiltration at the landscape level by restoring ecosystems and enhancing natural processes. Restoring these functions will improve long term stability of groundwater levels and result in less extreme fluctuation in river flow. We link this to three important adaptation challenges closely related to water management and hydrological functioning. These adaptation challenges are: 1) Sustainable water provisioning from groundwater resources 2) Base flow maintenance of surface hydrology and associated issues regarding navigability and water quality 3) Reduction of peak flows and associated flood risk to downstream. Challenges related to water quality, climate mitigation and biodiversity (conservation objectives) are not included in the prioritisation method, but are obviously important to bear in mind. Implicitly, EbA inspired restoration scenarios will probably affect a broad range of ES (e.g. recreation, health effects, fishery, etc.).

There are four key measures to increase the hydrological resilience to droughts that are put forward by the PROWATER project.

A: INFILTRATION RESTORATION THROUGH FOREST CONVERSION

Unmanaged forests can have high rainfall interception. Especially pine and fir have very high interception. Often such forests have been planted on the elevated and dry sites. These sites are also important for groundwater recharge as groundwater levels are deep and far from draining streams. Forest conversion to broadleaf forest or more open vegetation types allows Building up additional groundwater head during winter, which mitigates impacts of droughts.

B: RESTORE PERMANENT NATURAL WATER RETENTION

Wetlands have been drained in the past for agriculture, forestry and even recreational housing. These sites now have an almost permanent draining of seepage water. By reducing drainage these sites can be restored. Over time, the peat layer can act as a sponge in retaining water. During winter, when evapotranspiration is low, these sites can build up water stocks to be released during summer. For many sites, past activities have been abandoned, but drainage infrastructure is still present.

C: RESTORE TEMPORAL NATURAL WATER RETENTION

Without artificial drainage, temporary wetlands would occur at many locations in upstream (dry) valleys and landscape depressions. These sites are dependent on local seepage and run-off dynamics. Periods of excess precipitation can lead to temporarily waterlogged conditions. Most often these sites are drained, but under natural conditions delayed infiltration would take place when the groundwater levels naturally decline during spring. Instead if draining these sites, water should be retained locally until infiltration is achieved. Restoration of the hydrological functioning can be achieved through active management of the drainage system and/or installing retention ponds.

D: INFILTRATION RESTORATION THROUGH THE REMEDIATION OF SOIL COMPACTION (& SOIL SEALING)

Soil compaction, also known as soil structure degradation/degradation, is the increase of bulk density or decrease in porosity of soil due to externally or internally applied loads. Compaction can adversely affect nearly all physical, chemical and biological properties and functions of soil. Together with soil erosion, it is regarded as the "costliest and most serious environmental problem caused by conventional agriculture." Soil compaction causes a decrease in large pores (called macropores), resulting in a much lower water infiltration rate into soil, as well as a decrease in saturated hydraulic conductivity. Soil sealing by infrastructure (paved surfaces) has a similar impact but needs more technical remediation.

Although the principles behind these EbA-measures may be clear on a conceptual level, water managers and spatial planners struggle to implement these principles in a practical context. Restoration measures for hydrological functions are often planned on a case-by-case basis, without considering the hydrological processes that are the links between the different functions and that manifest themselves on a catchment scale (Palmer 2009; Horvath et al. 2017). This also implies that mostly only local benefits of the restoration are considered. However, more benefits could be achieved if restoration is addressed strategically on the catchment level (Horvath et al. 2017). Especially with challenges related to climate change, addressing issues at larger spatial scales becomes more prominent (Dragut and Blaschke 2006).

To be able to integrate these principles into spatial planning and management, tools are needed that can be applied on the catchment level and that allow to map the suit of restoration measures targeting different hydrological functions (e.g. infiltration, retention, …). Restoration measures can be scattered throughout the landscape, occupying only small areas at each site, but together have a strong influence on the hydrology of the entire catchment (Merot et al. 2006). For example, in several areas the ecologically most diverse wetlands have a size of less than 0,3 ha (Leonard et al. 2012; Semtlichs et al. 2015; Riley et al. 2017). It is therefore important that the tool has a sufficiently high spatial resolution so that these small-scale features can be identified. Especially in intensively used areas where available space is limited it is important to take efficient measures, or when funds are scarce (Walter et al. 2007; Wilson et al. 2011).

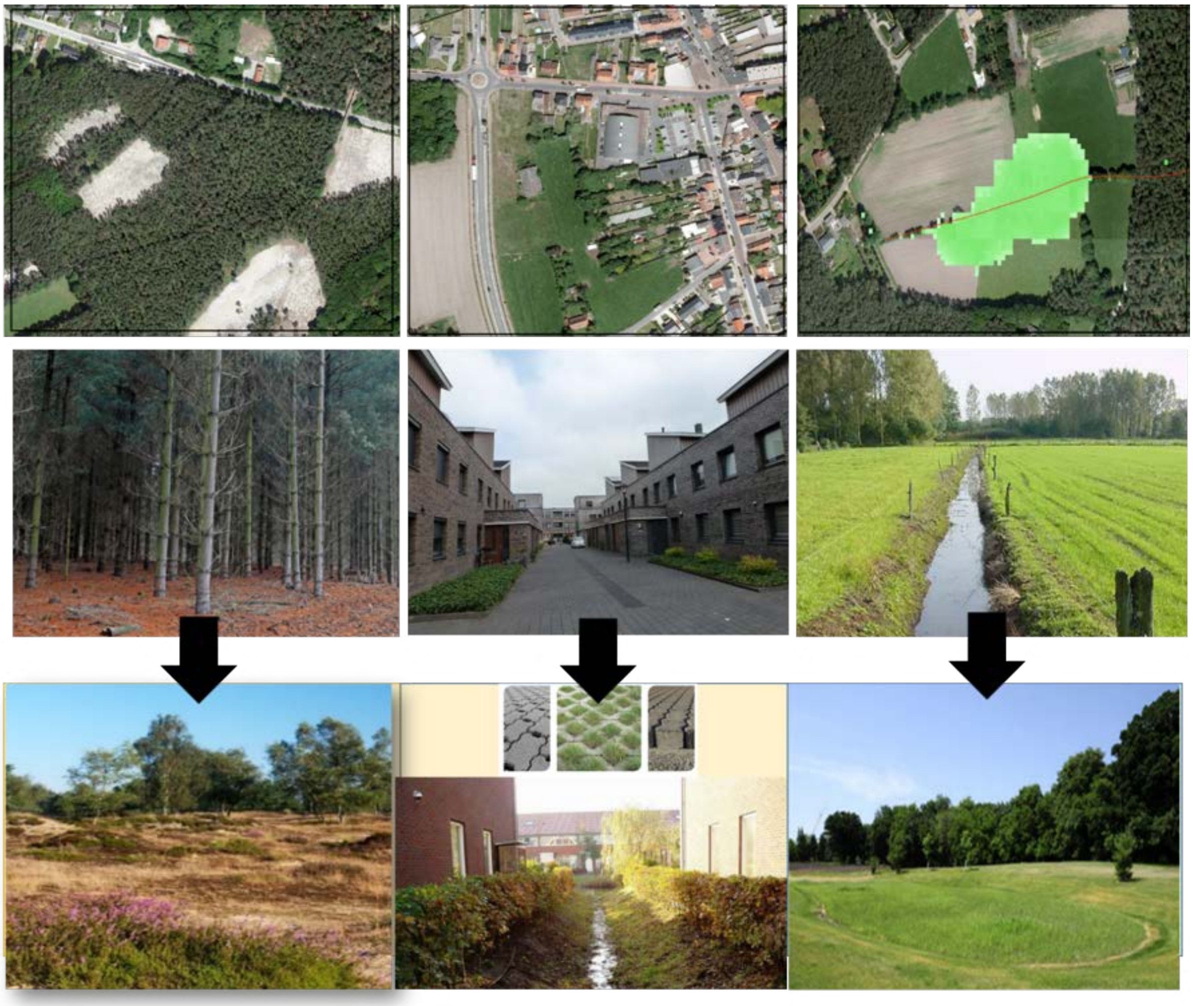


Figure 3: Examples of measures that improve retention and infiltration. Left: unmanaged pine plantations on dry elevated soils intercept a lot of precipitation in their dense canopy and thick litter layer. Middle: Paved surfaced cause soil sealing and in many cases run-off water is collected and piped to nearby streams. Instead, runoff water should be collected infiltrated locally in wide open ditches. Right: A natural landscape depression on a slowly permeable subsoil which is drained for agriculture. Drainage can be minimized by placing adjustable weirs and the lowest part of the depression can be restored to act as a buffer to allow deferred infiltration.

Implementing measures without a clear spatial vision can result in poor effectiveness. Especially when there are evident spatial processes that underlie the mechanisms of ecosystem service supply. Infiltration - retention patterns occur on various spatial and temporal scales and the actual contribution to the regulation services will depend on the topographic position at which the ecosystem function occurs. When these spatial-temporal processes are not considered, there is a risk that the residence time of the infiltrated or retained water will be too short to effectively mitigate a subsequent drought episode. Ideally, infiltration and retention are promoted on the most upstream elevated terrains that separates neighbouring drainage basins. In these zones, headwater wetlands form the source areas for many small streams.

An SPM for EbA-measures will provide guidance on where to implement particular infiltration/retentions measures to have the greatest benefit in terms of reducing drought risk per unit surface area. The application of principles and associated measures requires a thorough understanding of the hydrological (dis)functioning of catchments. Spatial indicators can be useful for the spatial and thematic prioritization and optimization of restoration measures at landscape-level.

Therefore, this deliverable provides a literature review of potential indicators and methods for spatial prioritisation. Based on these insights, the deliverable already puts forward a preferred PROWATER approach to spatial prioritisation of EbA measures and their preliminary validation.

2. Introduction to geomorphometry and scope of the review

One of the most commonly used variables in SPM's is topography. These methods are based on the hypothesis that relief primarily determines runoff flow patterns (Dosskey and Qiu 2011) and groundwater flow (Gallant and Wilson 2000), and that from there other functions can be derived, such as spread of invasive species (Murphy et al. 2015), sediment flow and accumulation (e.g. Renard et al. 1991), water purification (Tomer et al. 2009), primary production (e.g. Wondzell et al. 1996; Bonell and Molicova 2003), soil development (McBratney et al. 2003; Behrens et al. 2010), habitat development (e.g. Thompson et al. 2006) and several other ecological processes (Dragut and Baschke 2006). Also, for the reduction of drought risks, the type of ecosystem-based measures that would be most effective at a certain point depends on its relative position in the landscape. Areas that are relatively low in comparison with their environment (e.g. valleys and local depressions) are more prone to accumulate and retain water that flows into the depression, while relatively high areas such as ridges and crests are more prone to infiltration.

Most geomorphometric methods (defined by Pike et al. 2009 as 'methods that study the land surface or topography in a quantitative way') consider the topography of the neighbouring area of a pixel as a basis to study the landscape. They are also referred to as topographic indices (Newman et al. 2018) or digital terrain indices (e.g. Agren et al. 2014). In the past decades, the field of geomorphometry has strongly evolved due to the increasing quality and availability of fine-resolution digital terrain data (Dragut and Baschke 2006), such as derived through LiDAR (Light Detection and Ranging, laser altimetry). Many new algorithms and software to calculate topographic indices were developed, including TOPMODEL (Beven and Kirkby 1997; Walter et al. 2002), Zevenbergen and Thorne (1987), TAPES (Wilson and Gallant 1998), and ready-to-use GIS-tools became available such as the Topographic Position Index (Jenness 2006), TauDEM (Tarboton 1997), Terrain Analysis System TAS (Linsday 2005), etc.

The neighbouring area that is considered in topographic indices can be defined by the pixels adjacent to the central point, in which case it is called 'the local neighbourhood' (3x3 window). In case it includes the area further upstream or downstream a pixel, it is called 'the extended neighbourhood' (Wilson and Gallant 2000). Roughly two classes of topographic indices can be distinguished: absolute and relative (Newman et al. 2018). Absolute indices express the difference in height above or below a point or value of comparison in absolute values. An example is the topographic position index, which is the difference between a central point and the mean elevation in the neighbourhood around the point (Weiss et al. 2001), expressed in the same unit as the elevation data. Relative indices take into account the variability or range in elevation in the neighbourhood around the point, by normalizing the elevation to the variability or the range. Relative indices are often used when result using multiple neighbourhood sizes are used in the analyses as the difference in elevation becomes larger with neighbourhood size (Newman et al. 2018).

Topographic indices can be used as single tools (only elevation data), or in combination with other parameters such as soil properties (e.g. Van Soest and Williams 2001; Merot et al. 2006; Dosskey and Qiu 2011; Infascelli et al. 2013), climate (Merot et al. 2003), satellite derived vegetation indices (e.g. Wu et al. 2014), geology (Merot et al. 2003), land use or infrastructure (e.g. Uuemaa et al. 2018), ..., in which case the selection of additional parameters is driven by the object of the analysis. For example, when it comes to wetland restoration for water quality improvement, information on upstream land use would be required to know the amount of nutrients that flow into the wetland and determine its efficiency in nutrient removal (e.g. Merot et al. 2006; Dosskey and Qiu 2011). The usage of topographic indices as sole indicators of dynamic hydrological functions implies that certain assumptions are made which are not always true, and that non-linear effects may not be captured appropriately (Wilson and Gallant 2000). Numerous studies discuss how compound indicators and process-based models can more accurately identify landscape attributes and functions than approaches based solely on topographic variables (e.g. Wilson and Gallant 2000; Grabs et al. 2009; Qiu 2009; Infascelli et al. 2013). However, with increasing complexity and accuracy also comes a higher need for input data, appropriate software and knowledge on how to apply the software correctly. Detailed information on soils, vegetation and other parameters is often not available on regional or larger spatial scales. Thanks to technical evolutions in remote sensing (e.g. LiDAR), elevation data is now more widely available on large spatial scales and high resolution. Especially for identifying small-scale measures high-resolution data is important, as they may otherwise not be detected. Several studies that tested the accuracy of topographic indices by comparing with field data, topographic and historic maps (Merot et al. 2006; van Deventer et al. 2016; Riley et al. 2017; Uuemaa et al. 2018), conclude that in spite of the lower accuracy, topographic indices are very useful when it comes to landscape-scale identification of potential sites restoration or preservation sites.

Hence, this review specifically targets literature based on following keywords: hydrological functions of infiltration, groundwater recharge and water retention; restoration and conservation (of hydrological functions); landscape analysis, mapping, elevation and topographic indices.

3. Overview of specific topographic indices

3.1. Topographic Wetness Index TWI

One of the most commonly applied topographic index to study hydrological functioning of the landscape is the Topographic Wetness Index TWI (Beven and Kirkby 1979; Beven 1986), or also called the Compound Topographic Index (Moore et al. 1991):

$$TWI = \ln\left(\frac{a}{\tan\beta}\right)$$

a = upslope drainage area
ß = local slope

A high value for TWI is found in areas with high flow accumulation and flats slopes, low values represent areas that are more prone to infiltration (Patil et al. 2018). The index is mostly applied to identify (potential) wetland areas (e.g. Merot et al. 2006; Maxa and Bolstad 2009; Landmann et al. 2010; Na et al. 2013; Besnard et al. 2014; Gorvath et al. 2017), although it has also been used to identify infiltration areas (e.g. Ballerine 2017; Patil et al. 2018), areas suitable for agriculture (e.g. Buchanan et al. 2014) and landslides (Rozycka et al. 2016). Qiu 2009 and Giri et al. 2018 used an extended version of the TWI to identify the source areas where the runoff of pollutant carrying water to buffer zones or wetlands originates from. In a study on wetland occurrence in Sweden, it was found that best results for predicting wetland location using the TWI are obtained when smoothing the DEM. The use of a fine resolution DEM did not allow to distinguish between upland and wetland (Sørensen and Seibert 2007). This led Agren et al. (2014) to conclude that the TWI is not the best method for subtle differences in wetness along wetlands, streams and lakes, or by extension small seasonal pools and temporary wetlands. Especially in landscapes with small-scale human modifications such as drainage ditches and berms, it is important to be able to use a high resolution of DEM, as they have a strong impact on soil saturation (Dosskey and Qiu 2011).

The TPI index in its simplest form does not take into account the stream order of the catchment and the downstream slope (Infascelli et al. 2013). However, the functioning of a wetland strongly depends on its position in the catchment, the stream order and the time of inundation (Riley et al. 2017). The method developed by Lang et al. (2013) proved to be successful to predict areas of inundation during dry period by combining the TWI with a local terrain normalized relief map. It is assumed by the authors that it could also allow to map interannual variability in the wetness of an area, although this was not further tested.

Also, the TWI is based on the hypothesis that the hydraulic gradient of the subsurface flow is determined by the local slope of the relief. However, research demonstrates that this hypothesis does not always hold true in areas where the riparian zone is not or only gently sloping and subsurface flow is influenced by the stream water level (Burt et al. 2002; Vidon and Hill 2004). Infascelli et al. (2013) proposed a modified TWI index (the ordinated-climato-topographic index) that takes into account both stream order (represented by the volume of annual effective rainfall depth and the volume drained by the stream) and the local downhill difference in level to the stream to account for inverse flow in gentle riparian slopes. Although results show to be more accurate, the need to supply information on effective rainfall and weighting factors for the influence of upslope and downslope variable may impede usage of this index in data-scarce areas.

3.2. Depth-To-Water index DTW

The DTW is defined as "the least-cost depth or elevation difference (in metres) to the nearest open water locations such as the DEM-derived streams, lakes, pools, ponds, or shoreline where DTW is set to be 0" (Agren et al. 2014):

$$D_{TW} = \left[\sum \frac{dz_i}{dx_i} a\right] x_c$$

dz/dx = slope of a cell along the least-elevation path
i = a cell along the path
a = 1 when the path crosses the cell parallel to the cell boundaries and 1.414214 when it crosses diagonally
x_c = grid cell size (m)

The DTW was developed by Murphy et al. (2007). It proved to better represent field observations of wetlands than the TWI due to the consideration of downslope topography and hydrological conditions and dispersive flow in low-lying landscape positions (Murphy et al. 2009; Agren et al. 2014). An additional advantage over the TWI, is that the DTW does not require smoothing of the DEM and is thus less sensitive to scale. The amount of detail is only restricted by the accuracy and the resolution of the available DEM (Murphy et al. 2007, 2009; Agren et al. 2014). Hence, DTW is better at mapping small-scale variations of wet areas along larger wetlands and water bodies than TWI (Agren et al. 2014). Using the TWI, high wetness values are more restricted to discrete lines of flow accumulation within wet areas, while the DTW maps wetted areas as a whole (Murphy et al. 2009; Agren et al. 2014; O'Neil et al. 2018). However, DTW shows to more often result in false positive predictions of wetlands (wetland predicted in non-wet area) than TWI (O'Neil et al. 2018).

3.3. Topographic Position Index TPI

The Topographic Position Index TPI (Weiss 2001), or difference from mean elevation (De Reu et al. 2011), calculates the difference between the elevation at a central point with the mean elevation in a given neighbourhood around the point:

$$TPI = z_0 - \bar{z}$$
$$\bar{z} = \frac{1}{n_R} \sum_{i \in R} z_i$$

z_0 = elevation at the central point
$\bar{z}$ = the average elevation around the central point within a predetermined radius (R)

The TPI is used to map landforms and functions at different positions of the landscapes, such as valley bottoms, hilltops and slopes. A positive value is found in areas that are elevated in comparison with their neighbourhood, negative values are found in depressions. In some studies, the median value is used instead of the mean, making the outcome less sensitive to outliers (Newman et al. 2018). However, as the highest and lowest values are smoothed out, rapid changes in local topography may also be masked out. In the research of Newman et al. 2018, this smoothing resulted in poor performance of TPI in the lowest and highest topographic conditions. The range of the TPI depends on the difference in elevation within the specified neighbourhood. The larger the neighbourhood, the more likely the range in elevation is large and so will be the TPI. A smaller neighbourhood will therefore be able to identify small-scale landscape features, while a large neighbourhood will show major landscape features (De Reu et al. 2011). In contrast to the TWI, the TPI takes into account downstream relief and is therefore able to position a feature within the larger landscape.

One of the shortcomings of the TPI according to Dragut and Blaschke (2006) and De Reu et al. (2013), is the strong influence of surface roughness on the outcome. Consequently, the TPI will not be able to distinguish small-scale landscape features in areas with a large variation in elevation. Especially in heterogeneous areas where both subtle (height differences of a few meters) and more pronounced topography are present, this may result in inaccurate outcomes. Dragut and Blaschke (2006) and De Reu et al. (2013) also point out that most TPI-based analyses focus on a single landscape feature, leading to under- or overrepresentation of certain landscape features in many studies (e.g. Van Deventer et al. 2016; Gruber et al. 2017).

Van Deventer et al. (2016) found that the usage of TPI to identify landforms on country-wide scale results in an overall accuracy of only 43%. Although the results reflected the diversity of the landscape, slope classes and plains were not well mapped and extremely low levels of accuracy were obtained in some catchments.

A major advantage of the TPI in terms of mapping hydrological functioning is its simplicity and straightforwardness in relating it to the duration of soil saturation (hydroperiod) of wetlands. The larger the difference in elevation between a point in a depression and its environment, the more water will flow to the pixel, the longer the hydroperiod and the less prone the wetland is to desiccation (Riley et al. 2017). The likeness that the relief intersects the groundwater table increases with elevation difference, thus providing a more stable supply of water to the wetland and longer saturation time. This allows to identify and map the type of wetland occurring in the landscape (upstream depressional wetlands, headwater wetlands or valley bottom wetlands). Likewise, the larger the difference in elevation between a point on a hilltop and its environment, the deeper the groundwater table at that location and the more likely the area functions as an infiltration zone for groundwater replenishment in case no impermeable layer is present.

3.4. Deviation from mean Elevation DEV

The DEV index is a derivative from the TPI. DEV normalizes the TPI by dividing it by the standard deviation of the elevation:

$$DEV = \frac{z_0 - \bar{z}}{SD}$$

$$SD = \sqrt{\frac{1}{n_R - 1} \sum_{i=1} (z_i - \bar{z})^2}$$

z_0 = elevation at the central point
z- = the average elevation around the central point within a predetermined radius (R)
SD = standard deviation of the elevation

The outcome is a z-score instead of an elevation unit, with a positive and negative value indicating that the central point is above and below the mean elevation within the neighbourhood (Newman et al. 2018). By normalizing the local relief to local surface roughness, the index becomes less sensitive to surface roughness in comparison with the TPI (less sensitive to outliers), allowing to identify both pronounced and more subtle landscapes features in a heterogeneous landscape (De Reu et al. 2013). However, by normalizing the topographic position, the DEV implicitly assumes that the elevation values in the neighbourhood are normally distributed (Newman et al. 2018). The likeliness that this is true increases with increasing size of neighbourhood, as more elevation values are present in the sample but for small-scale analyses this may not be true and produce inaccurate results.

3.5. Elevation Percentile EP

The EP ranks the elevation of the central point compared to all points within a user-defined ratio (Wilson and Gallant 2000):

$$EP = count_{i \in C}(z_i > z_0) \times (100/n_c)$$

z_0 = elevation at the central point
z_i = elevation of cell i contained within the neighbourhood C
n_C = number of cells within the neighbourhood C

The index EP developed by Gallant (1996) ranges between 0 and 100%, where low values represent upper-slope positions and high values represent wetter areas in low topographic positions. A comparison of 4 topographic indices (EP, DEV, PER, RTP) revealed that EP provides the most robust results for geomorphometric terrain analysis (Newman et al. 2018). This is explained by the insensitivity of the index to outliers in the DEM and to irregularly shaped elevation frequency distributions. However, the calculation of EP requires much more time (up to 225 times more than other indices according to Newman et al. 2018). This may reduce applicability of the index when using high-resolution DEM's such as provided by LIDAR (De Reu et al. 2011). Advances in image processing and computer visioning, such as integral image approach, can help to reduce calculation times when applying topographic indices on multiple scales, allowing to combine both robustness and time-efficiency (Lindsay et al. 2015; Newman et al. 2018). EP is a measure of local topographic position and is sometimes used to study spatial patterns in vegetation cover (Wilson 2018).

3.6. Percent Elevation Range PER

PER measures the position of a pixel above the minimum elevation within its neighbourhood and is expressed as a percentage of the elevation relief (Newman et al. 2018):

$$PER = (z_0 - z_{min})/(z_{max} - z_{min}) \times 100$$

z_0 = elevation at the central point
z_{min} = minimum elevation within the neighbouring distribution
z_{max} = maximum elevation within the neighbouring distribution

PER is expressed as a percentile and is straightforward to calculate in GIS software. Main advantages of the index are its straightforwardness in interpretation, its possibility to calculate it with any (spatial) software that has a tool for minimum and maximum filtering and the speed with which the calculations can be performed. A disadvantage of PER is that it assumes a linear relationship between the minimum and the maximum elevation value and ignores different shapes of the elevation distribution. Hence, results of PER calculations have relatively low accuracy in comparison with other metrics (Newman et al. 2018). The index may be suitable for explorative purposes (quick scan) but is inadequate to identify areas for restoration of hydrological functions.

3.7. Relative Topographic Position index RTP

The RTP was developed as a modification of PER to be able to take into the shape of the elevation distribution (Newman et al. 2018). Besides minimum and maximum elevation, it also positions the elevation at the central cell relative to the mean elevation as follows (Newmand et al. 2018):

$$RTP = \begin{cases} \dfrac{z_0 - \mu}{\mu - z_{min}}, if\ z_0 < \mu \\ \dfrac{z_0 - \mu}{z_{max} - \mu}, if\ z_0 > \mu \end{cases}$$

z_0 = elevation at the central point
z_{min} = the minimum elevation in the neighbourhood around the central point
z_{max} = the maximum elevation in the neighbourhood around the central point
μ = the mean elevation in the neighbourhood around the central point

The RTP varies between -1 (central cell is below mean elevation) and 1 (central cell is above mean elevation). The RTP reveals to be more accurate than PER (Newman et al. 2018). However, like the PER, RTP remains sensitive to outliers due to the usage of the minimum and maximum. The use of the mean also implies that a normal distribution is assumed, which may not always hold true when the index is calculated in small areas and result in false predictions of wet and dry areas. For sufficiently large areas the index proved to be one of the most robust topographic indices (Newman et al. 2018).

4. Importance of multi-scale assessments

Because topographic indices compare the elevation at one point with its surroundings, outcomes are strongly dependent of the size of the chosen neighbourhood (Chang and Tsai 1991; Deng et al. 2007; Sørensen and Seibert 2007; Lindsay et al. 2018; Newman et al. 2018). Topographic analyses using a small neighbourhood will reveal minor landforms such as small wetlands in headwater catchments or seasonal pools while a large neighbourhood will reveal major landforms (e.g. land dunes). Choosing the most appropriate scale for the analysis requires expertise and understanding of geomorphometry (De Reu et al. 2011). However, the availability of tools and plugins for the calculation of topographic indices in common GIS software packages such as ArcGIS or QGIS has led to uncritical application of the tools by users without proper background (De Reu et al. 2013). Also the resolution of the digital elevation map will have a strong effect on the results (Gallant and Dowling 2003; Deng et al. 2007), as differences in elevation of the central point become smoothed out with larger resolution, potentially resulting in inaccurate representation of the terrain (Wood 1996). This further reduces the objectivity with which topographic analyses are performed (Riley et al. 2017) and risks inaccurate results when geometric experience or understanding is lacking (De Reu et al. 2013). Several approaches have been proposed to deal with scale-variant results of topographic analyses, such as combining DEM's of different resolutions into a single index (Gallant and Dowling 2003) or using a hierarchical object-delineation approach (Dragut and Eisank 2011). One of the most straightforward ways is to combine analyses at different scales into a single index, also referred to as varying roving window sizes or raster spatial filtering (Lindsay et al. 2015). Grohmann and Riccomini (2009) conclude that this method should be best restricted to analysis at small scales due to computational intensity on larger spatial scales. On the other hand, Behrens et al. (2010) recommend in general to use a multi-scale approach to topographic index calculations to improve accuracy of the outcomes. According to O'Neil et al. (1986), at least three scales would be required for correct topographic analyses. Image-based transformation is a promising technique to combine analyses of a large number of neighbourhood scales in a highly computational-efficient way (see e.g. Lindsay et al. 2015).

Multi-scale approaches have several additional advantages. Topographic analysis at one scale provides information on topographical characteristics, but also the difference in the results at multiple scales provides complementary information (Dragut et al. 2011; De Reu et al. 2013; Newman et al. 2018). This variation of topographic index values over a range of scales is also known as the scale signature (Wood 1996; Lindsay et al. 2015). Combining calculations at different scales allows to identify landforms (Weiss 2001), and these landforms (e.g. wetlands, hillslopes, ...) each have their own hydrological behaviour and ecological functioning (e.g. infiltration, gully erosion, water purification, ...). The larger the difference in the topographic indices between divergent scales in a given cell, the more variable the groundwater level. This may reveal seasonal or weather patterns in groundwater level. For example, for TPI, areas with a high TPI in a small neighbourhood and a low TPI in a large neighbourhood have a small contributing area and receive less runoff and groundwater during rainfall events compared to areas with high TPI in large neighbourhood. Their groundwater level will therefore fluctuate more with the seasons and with extreme events. These differences among the TPI at multiple scales within a cell reveal information on the hydroperiod of wetlands and allow to distinguish between different types of wetlands. According to the landform classification of Weiss (2001), for example, these areas with high TPI in a small neighbourhood and low TPI in a large neighbourhood would be identified as upland depressions (Jenness 2006).

5. PROWATER approach

5.1. Key principles behind the PROWATER approach

At the core of the PROWATER SPM, we make use of the topographic position index (TPI). The TPI can be calculated as the difference between the elevation at one central point and the mean elevation within the neighbourhood around that point (Gallant and Wilson 2000; Weiss 2011). Therefore, the TPI is a scale-dependent indicator and determines the relative position of each pixel, compared to the surrounding pixels within the landscape. The "surrounding pixels" are defined by setting a spatial distance (radius) to which the position of a pixel is compared (figure 4). The TPI has been used by Jennes (2006) to develop a simple and repeatable method to classify the landscape into slope position and landform categories. Compared with other topographic indices (discussed in Chapter 3), the TPI and derivatives such as the DEV have the advantage to be straightforward in interpretation on hydrological functioning while providing robust results (De Reu, Bourgeois et al. 2013; Newmand et al. 2018).

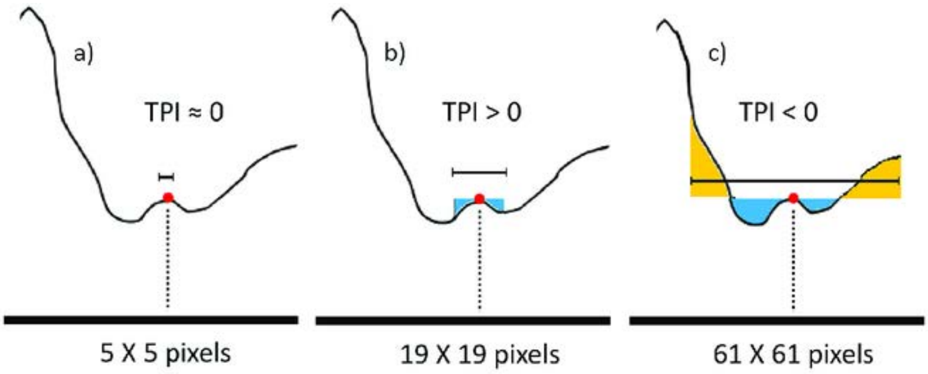


Figure 4: Depending on the neighbourhood, the relative position of a pixel can be very different. You may find yourself on an elevated part at the small scale (TPI > 0), but equally in a landscape depression at larger scale (TPI < 0) (taken from Jennes, 2006).

An advanced pre-and post-processing was developed to make optimal use of the strengths of the TPI. A pre-processing attempt to "abrade" the topographic elevations without filling up the depressions, mimicking a shallow groundwater layer beneath the surface. Small topographic elevations are erased from the surface. Because the range of TPI values depends on the scale (that is the window size of the neighbourhood around a central pixel for which the TPI calculation is performed) of the analysis, a post-processing is applied to standardize the TPI values into a fixed range and legend to be able to combine TPI's at different spatial scales (referred to as the deviation from mean elevation DEV in Chapter 3). When analyses are carried out at different scales, it is better to normalize the data, because the difference in height for a small-scale analysis will be smaller than for a large-scale analysis. This allows to apply the tool both to both subtle as well as to more pronounced topography (De Reu et al. 2013).

1. A low pass filter is applied on the DEM. The radius of the low pass filter is equal to the TPI radius that is used in step three.
2. The low-pass values are then used to overrule the original DEM values at locations where the filter values have a lower elevation then the original DEM (ridges are removed while depressions are not filled).
3. The TPI values are calculated on the defined spatial radius.

4. For that same spatial radius, we calculate the mean TPI and standard deviation.
5. We then standardize the TPI values by subtracting the mean TPI and dividing it by the standard deviation.
6. Finally, we apply a slice operation for the whole map, by which TPI values are reclassified to match an equal area distribution. This results in a map where each class (0-100) represents 1 % of the map area.

This procedure thus allows mimicking subsurface flows, by removing topographic elevations without filling up depressions. The advantage of this procedure is that the final map can be interpreted as a gradient of wet-dry soils (%) for a user defined spatial scale. When this procedure is applied on a range of scales (figure 5), we can unravel the interplay of infiltration-seepage relations.

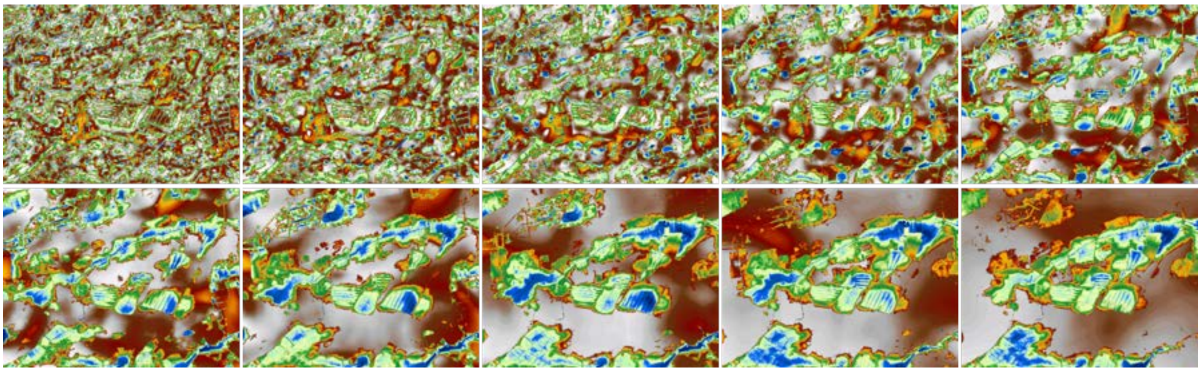


Figure 5: Infiltration seepage patters, calculated with the PROWATER method. For each spatial scale the infiltration seepage patterns are calculated. At the top-left side we see small scale patterns (infiltration-seepage within a 100-meter radius) and at the right bottom we see large scale patterns (infiltration-seepage within a 2500-meter radius).

A "gradient" of infiltration to seepage (resp. run-off generation and collection) can be calculated for a range of spatial scales. For each "spatial scale" an index (0-100) is calculated that represents infiltration-seepage patterns (0 = seepage, 100= infiltration). Each class corresponds to 1% of the area that has been analysed. We can then assume that roughly 60 % of the area has distinct infiltration potential and 10 % has distinct seepage properties (runoff collection). The remaining 30 % of the area (between 11 and 40) has intermediate properties and is not particularly wet or dry.

As the aim of the PROWATER project is to restore and protect different hydrological functions, various scales are of relevance. By combining multiple scales, landscape features at different scales and for different functions can be identified. The multiscale image method, as was developed by Lindsay et al. (2015), demonstrates how such a multiscale TPI approach can be used to classify landscapes into 8 distinct categories. Because the aim of the PROWATER project is to develop a generic tool that can be applied in different parts of the 2 Seas region, we make use of the same principles to distinguish landforms with distinct hydrological functions, e.g. permanent wetlands, temporary wetlands, retention areas, ...

Depending on the context (soil/topography/geology) and the type of interventions/measures, the user can focus on either small-scale processes or more large-scale processes. On heavy soils or soils with impermeable layers, a small scale is useful to detect runoff collection areas and potential water storage zones. A large-scale analysis might be less useful because large scale groundwater flow patterns are less relevant. But for catchments with sandy soils and permeable sub-soil, a large-scale analysis size may be very informative to detect temporary and permanent wetlands (groundwater seepage areas). To make these 'scales' more comprehensible, we defined a micro, meso and macro scale by assigning a minimum and maximum spatial range for TPI calculations. For each scale, a composite TPI map was calculated (mean TPI value for a set of spatial ranges). The PROWATER SPM also allows the user to store the individual maps as composite multiband layers for each scale.

Table 1: Definition of spatial scales for macro-meso-micro and their application potential

DEM	Macro	Meso	Micro
Pixel size	10m by 10m	5m by 5m	1m by 1m
Range (pixels)	100, 150, 200, 250, 300, 350, 400, 450, 500	20, 30, 40, 50, 60, 70, 80, 90, 100, 120, 140, 160, 180, 200	10, 20, 30, 40, 50, 60, 70, 80, 90, 100 m.
Range (meters)	1000, 1500, 2000, 2500, 3000, 3500, 4000, 4500, 5000 m.	100, 150, 200, 250, 300, 350, 400, 450, 500, 600, 700, 800, 900, 1000 m.	10, 20, 30, 40, 50, 60, 70, 80, 90, 100 m.
# Maps	9 map composite	14 map composite	10 map composite
Application	Identification of permanent wetlands (permanent seepage) and floodplains.	Identification of temporary wetlands. Detection of potential water storage (landscape depression).	Identification of runoff collection areas, small ponds and (drainage) ditches.

It is evident that the cell size of the DEM determines the scale of analysis. Although we advise to use the proposed resolutions, the micro, meso and macro scale can be calculated for a DEM with a finer of coarser DEM-resolution. Nonetheless, the differences in DEM-resolution for the macro-meso-micro scale analysis should remain the same. For the micro scale, the Cell Factor should be set to 1, for the meso scale, this should be 5 and for the macro scale this should be set at 10 (or 2 when using the DEM created by the meso scale calculations).

5.2. Macro scale patterns (1km – 5km)

The macro scale patterns are calculated using a 10by10 meter DEM. For several radii (radius of resp. 100, 150, 200, 250, 300, 350, 400, 450, 500 pixels) the "PROWATER" TPI procedure is applied. The 10 maps correspond to the topographic position of a pixel compared to the surrounding elevation within a range of 1 to 5 km. These 10 maps are combined into a mean TPI for large scale patterns. Macro Scale Composite = equal weighted sum of Radii 100, 150, 200, 250, 300, 350, 400, 450, 500 pixels. This allows to detect wetlands with strong seepage that are permanently wet throughout the year.

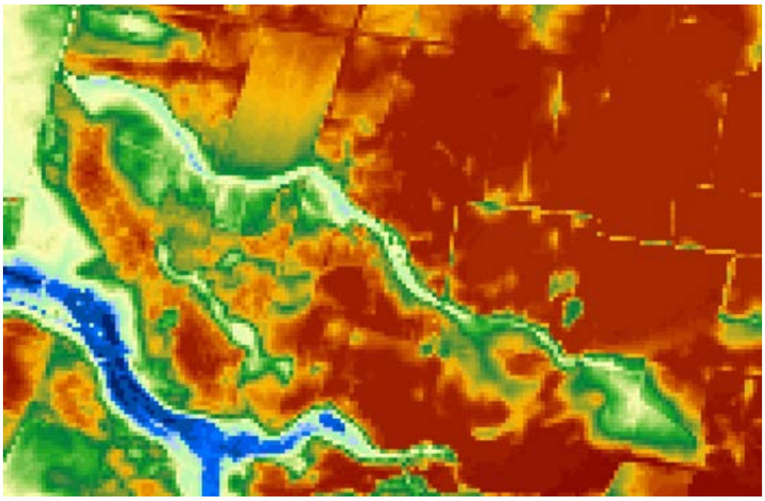


Figure 6: illustration of the macro-scale TPI

5.3. Meso scale patterns (0,1km – 1km)

The meso scale patterns are calculated using a 5by5 meter DEM. For several radii (radius of resp. 20, 30, 40, 50, 60, 70, 80, 90, 100, 120, 140, 160, 180, 200 pixels) the "PROWATER" TPI procedure is applied. The 14 maps correspond to the topographic position of a pixel compared to the surrounding elevation within a range of 100 to 1000 m. These 14 maps are combined into a mean TPI for small scale patterns. Meso Scale Composite = equal weighted sum of Radii 20, 30, 40, 50, 60, 70, 80, 90, 100, 120, 140, 160, 180, 200 pixels. This allows to detect wetlands with smaller recharge areas that are only temporarily wet after periods with high precipitation surplus.

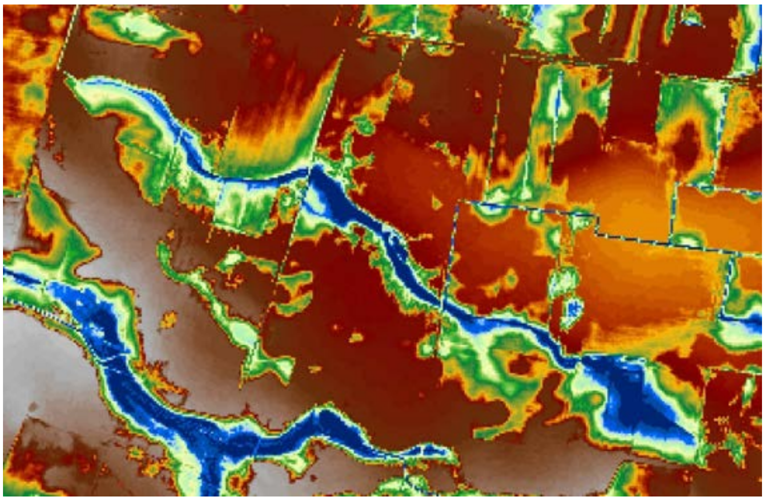


Figure 7: illustration of the meso-scale TPI

5.4. Micro scale patterns (10m – 100m)

The micro scale patterns are calculated using a 1by1 meter DEM. For several radii (radius of resp. 10, 20, 30, 40, 50, 60, 70, 80, 90, 100 pixels) the "PROWATER" TPI procedure is applied. The 10 maps correspond to the topographic position of a pixel compared to the surrounding elevation within a range of 100 to 1000 m. These 14 maps are combined into a mean TPI for small scale patterns. Micro Scale Composite = equal weighted sum of Radii 10, 20, 30, 40, 50, 60, 70, 80, 90, 100 pixels. This allows to detect where runoff water converges in the field. It also allows to detect streams, pond and ditches when the lowest values are selected (1-5).

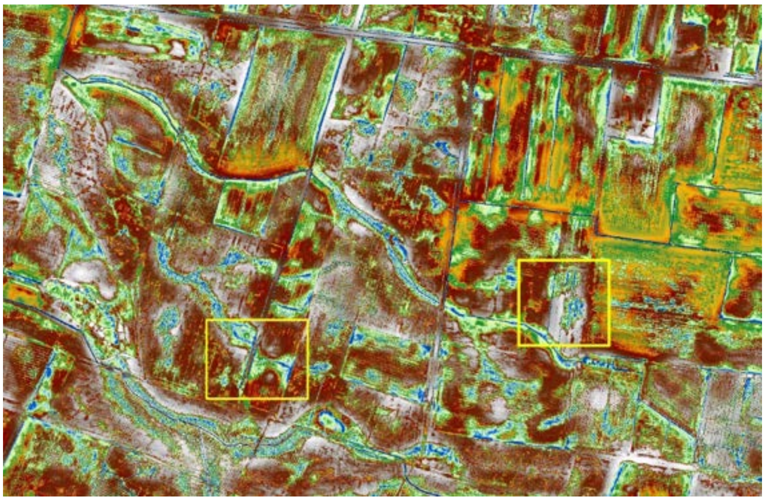


Figure 8: illustration of the micro-scale TPI

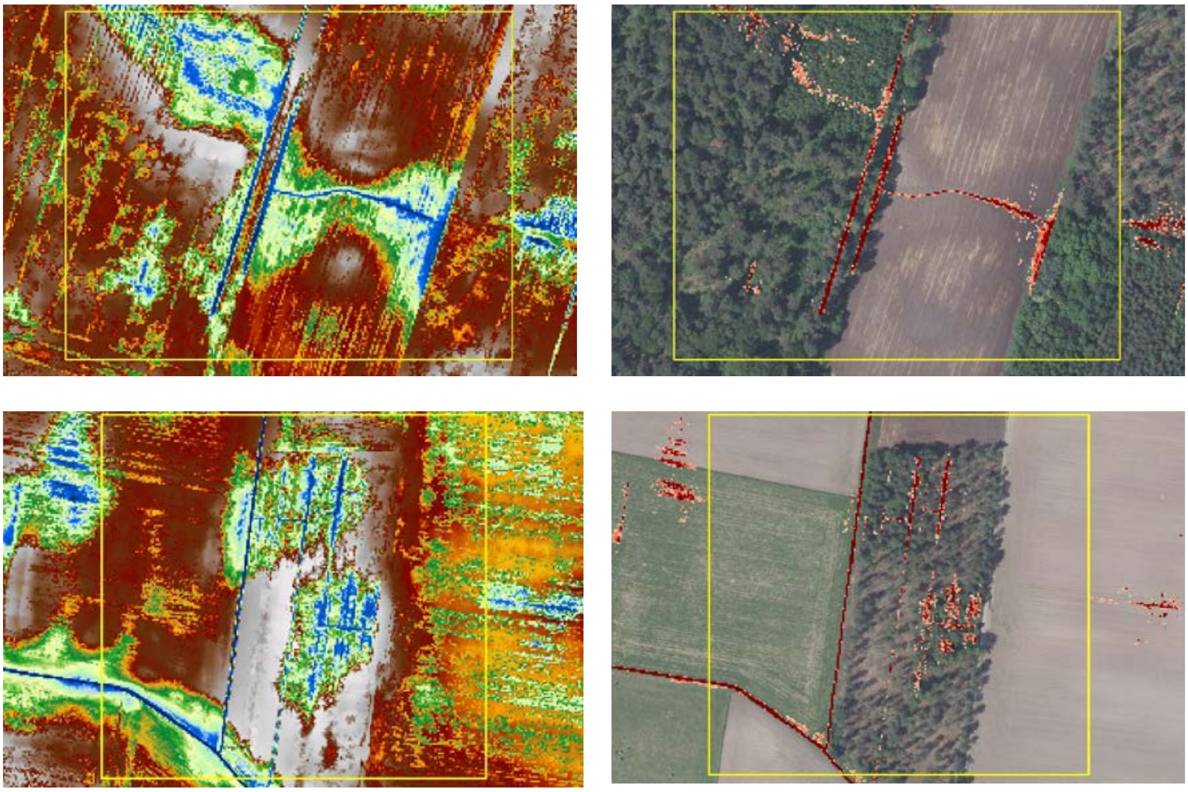


Figure 9: Visualisation of field level runoff collection areas and (drainage) ditches.

5.5. Defining macro-meso-micro interactions

Many sites that have low TPI values (relative landscape position is low = wet conditions) at the macro scale, also have low values at the meso and/or micro scale. While other sites only have a low TPI value at the meso scale and not at the macro scale. From a management perspective It is useful to distinguish them because the latter have a much higher potential for climate adaptation. Therefore, it is interesting to combine the different scales.

At the macro scale we can distinguish the infiltration seepage patterns that identify the main recharge areas and (potential) permanent wetlands. These wetlands receive a stable and permanent groundwater flow from upstream catchments (C type wetlands, Figure 10). At smaller scales we can observe more differentiated infiltration seepage pattern. Especially in upstream catchments we can detect the (temporary) headwater wetlands (B) and landscape depressions (A). Because the recharge areas for these systems are relatively small, there is a strong seasonal effect of the seepage intensity. At the smallest scale we can identify the field level landscape depressions and drainage ditches that collect run-off.

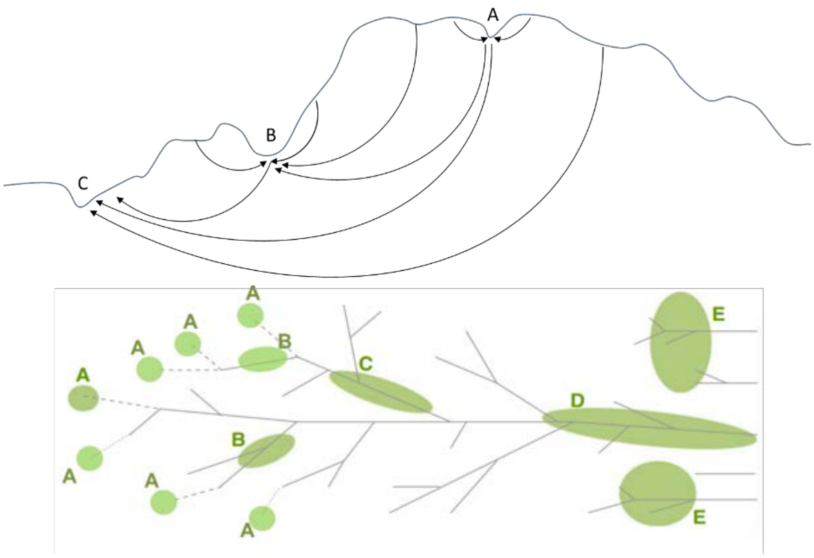


Figure 10: Interplay of infiltration-seepage relations within a landscape. Wetland type A: Upstream Depressional Wetlands (fens) are landscape depressions that are temporarily filled with runoff and local interflow. Wetland type B: Headwater wetlands that deliver base flow to the formation of small streams. Wetlands Type C: valley bottom wetlands with permanent groundwater seepage. Wetland Type D: Floodplain wetlands of large rivers. Wetlands type E: Coastal wetlands under the influence of tidal regimes.

The role of upstream (small-scale) temporary wetlands in catchment functioning has been largely overlooked, primarily because such small wetlands (< 1 ha) fall outside the remit of most wetland inventories and are considered problematic to manage (Merot et al., 2006; Cohen et al., 2016; Evenson et al., 2018) and these wetlands are also too small to assess with conventional catchment scale hydrological models. Small wetlands also have often been viewed as problematic in terms of agricultural production and, consequently, have been subject to land drainage or infilling (Acreman & McCartney, 2009).

Based on an initial analysis for the Campine Region (Belgium), we can indeed conclude that a very large proportion of the upstream landscape depressions actually form the starting point of drainage networks. It is clear that artificial drainage channels have been created to connect topographically isolated fens to the hydrological network. It is also striking that this is not only for agricultural land, but also for forested areas. Drainage of these upstream landscape depressions may have an important impact on the replenishment of deep groundwater.

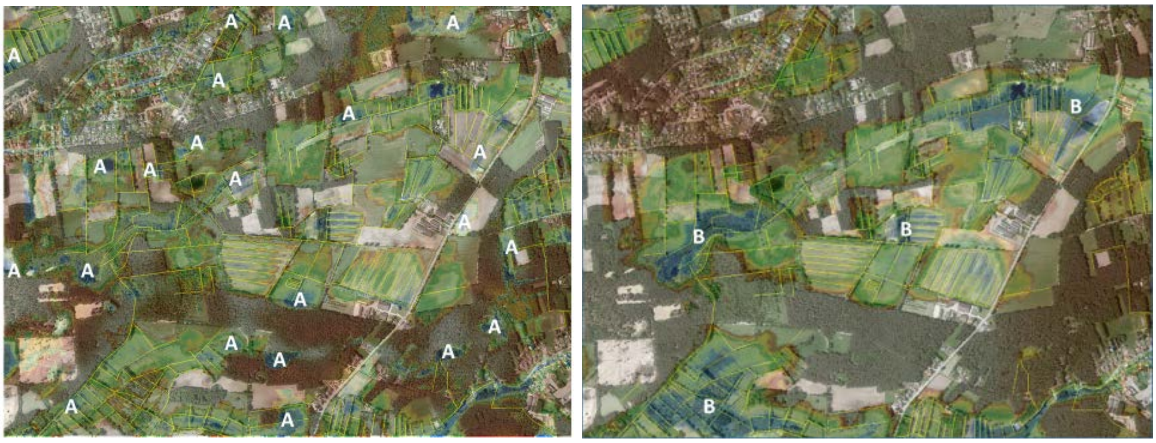


Figure 11: Cross-section of a landscape in the Campine Region (Belgium), on which we have indicated the many small upstream depressional wetlands (A). In 90% of the cases these landscape depressions form the starting point of a drainage ditch. It is precisely these many small landscape depressions in the capillaries of the water system that we need to rewet.

These widespread practices help to explain why even in rural areas, we only need a few days of abundant rainfall to get high water levels in rivers and ditches. Because we discharge this shallow ground water prematurely, it does not have the opportunity to infiltrate properly. If our changeable weather evolves to a situation where the precipitation is even more concentrated in time, we will lose even more groundwater recharge.

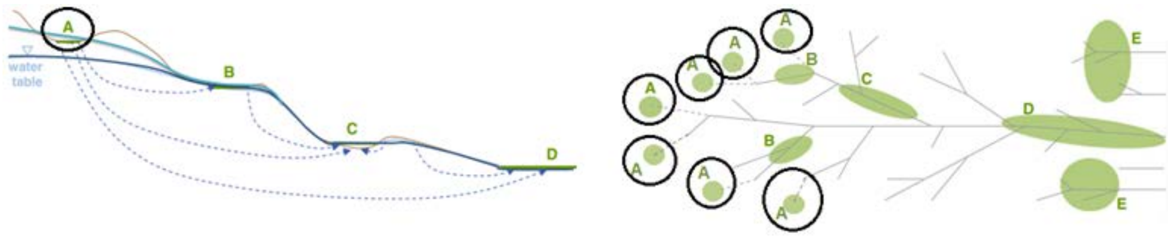


Figure 12: Wetlands along a climate and drainage gradient (Fan & Miguez-Macho; 2011). We focus on wetland types A and B which have the highest potential for climate adaptation. Many B type wetlands have been drained by enlarging and deepening natural streams. In addition, many hydrologically isolated A-type wetlands have been connected through artificial drainage ditches.

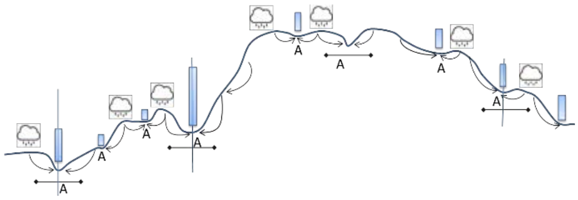
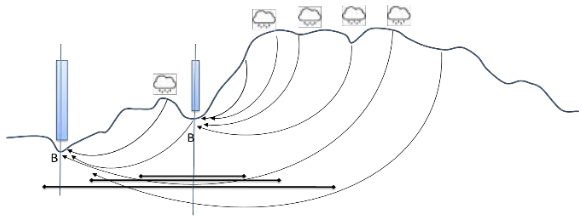
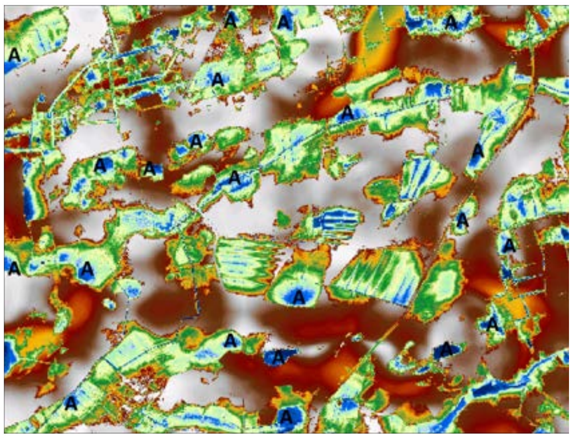
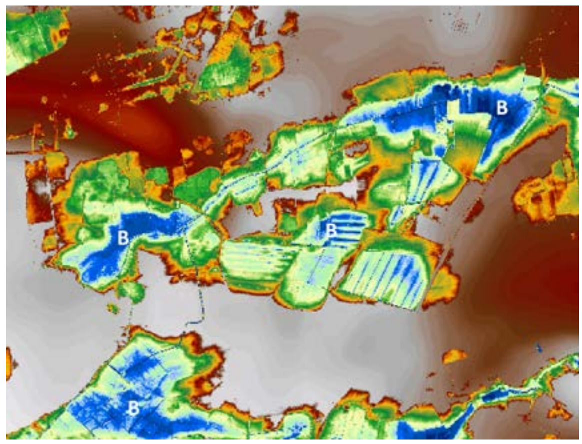
Relative position in the landscape at MESO scale (100 – 1000 m)	Relative position in the landscape at MACRO scale (1-5 km)
	
	
Blue-green colors indicate local depressions in the landscape where water accumulates.	Blue-green colors indicate headwater wetland areas on higher spatial scale.
When not drained, temporary wetlands occur with water levels that react fast to precipitation surplus. Generally wet conditions during spring, but increasingly drier conditions in the growing season. These wetlands are too small to support a base flow and are naturally not connected to the river network.	These sites have more permanent seepage, depending on the observed spatial scale. For the illustration above, the blue zones can be considered as headwater wetlands, which can support a small, but permanent water supply to small streams. Because of permanent wet conditions, peat formation is possible.

Figure 13: Spatial identification ad system functioning of meso-scale and macro-scale wetlands.

Many sites that are identified as B-type wetlands also have A-type wetland properties and it is useful to distinguish them from a management perspective. Therefore, it is interesting to combine the small-scale analysis with a large-scale analysis.

The 5 % pixels with the lowest value for the large-scale TPI are selected as permanently wet (steel blue). Within these zones, we further differentiate the seepage intensity, using the small-scale TPI. We give these zones a gradation of seepage intensity (0-25) and therefore this also reflects a gradual importance for water conservation measures. In these areas, unnecessary drainage should be avoided and thus certainly safeguarded from urbanisation.

The 25 % pixels with the lowest small-scale TPI that do NOT coincide with the 5 % wettest large-scale areas are considered to be temporarily wet and therefore have potential for deferred infiltration (green-blue). We give these zones a gradation (0-25) that indicates how important it is to retain water. The highest value corresponds to the lowest zones within landscape depression. These zones should not be built on. Here, too, it is best not to drain. These are the zones where runoff water from the surrounding area can be collected and retained.

For all zones that do not belong to the previous categories, we calculate an index for infiltration suitability (brown). By multiplying the large-scale and small-scale TPI, we obtain a combined infiltration suitability. The higher the value (0-100), the more suitable for infiltration. From the point of view of groundwater replenishment, it is best to select zones with a value higher than 40 (dark brown). Zones with a score lower than 40 are likely to be close to (potentially drained) wetlands and have rather shallow groundwater (between 50-100 cm).

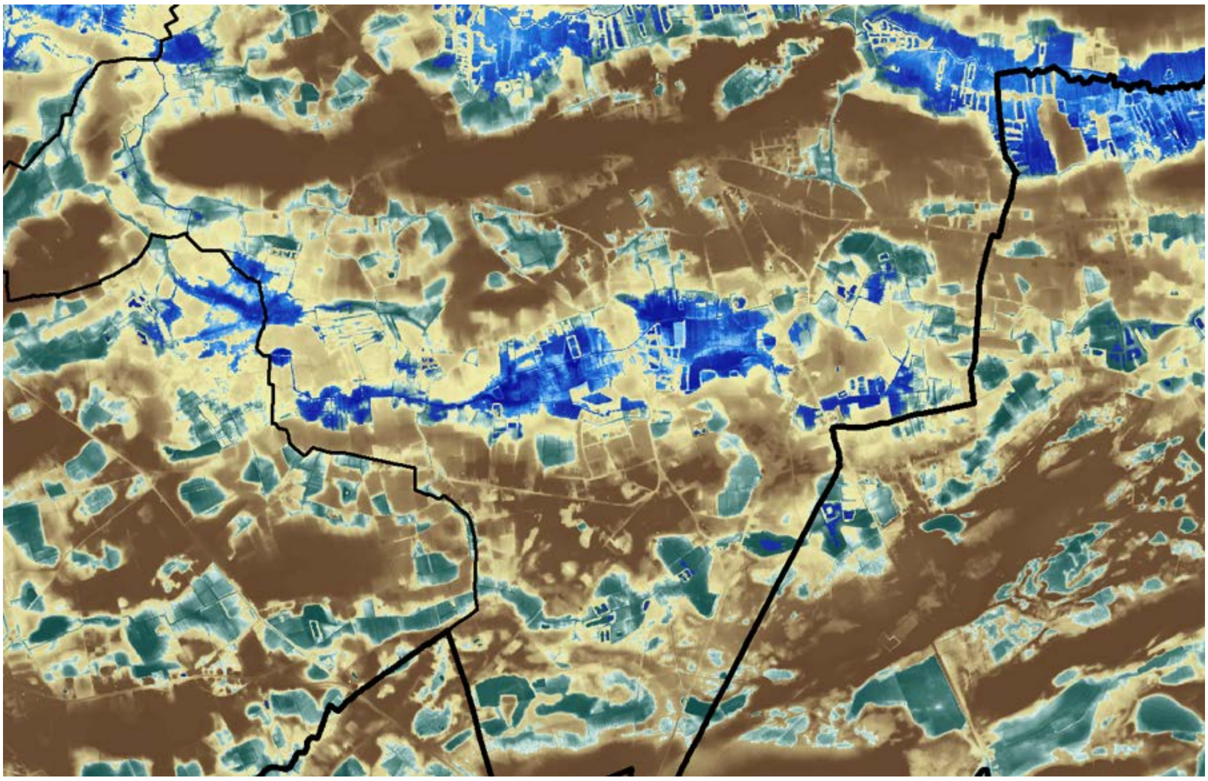


Figure 14: Water system map in which we distinguish between permanent wet seepage areas (steel blue, gradation in 25 classes), temporary wet zones (green blue, gradation in 25 classes) and drier infiltration zones (brown, gradation in 100 classes).

Such a landscape analysis (and classification) allows to differentiate where particular measures are more effective in achieving hydrological resilience. To achieve hydrological resilience, we need to enhance groundwater recharge (infiltration) at locations that are topographically elevated at larger scales. But this should we combined with a strategy to reduce the drainage of temporal wetlands. Within the topographically elevated areas, there are still a lot of small-scale landscape depressions and headwater wetlands. Retaining water in these upstream wetlands allows to build a water stock that can slowly infiltrate and/or provide base flow in late spring and summer.

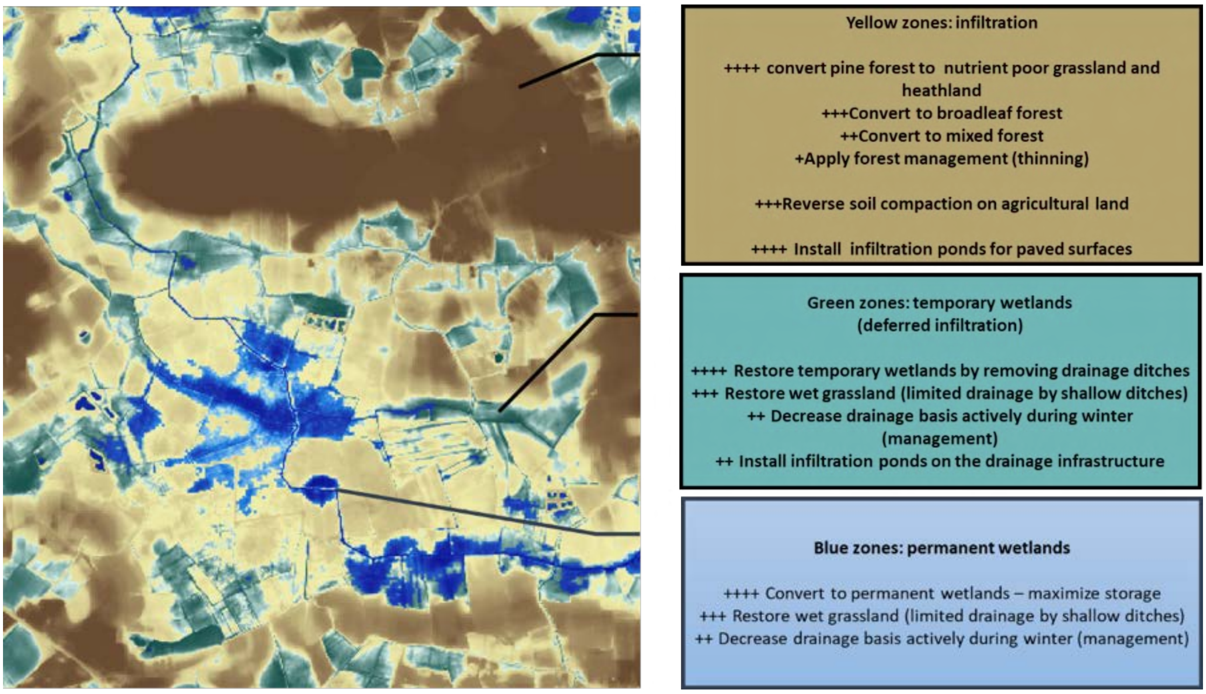


Figure 15: measures that we can associate with the water system map.

To build resilience against droughts, we primarily focus on direct infiltration and deferred infiltration through retention (brown and green zones). Depending on the current land use, there are several options to restore infiltration and retention. Figure 3 shows a few illustrations of measures that can be taken for very different land uses (forestry, peri-urban & farmland).

6. Validation results

The PROWATER SPM has been validated by applying it in 38 sites in three different countries (Netherlands, UK, Belgium). Table 2 gives an overview of the validation sites. The validation of the SPM is not finalized, but these results give a preliminary insight in the application potential of the SPM.

Table 2 – Overview of the test sites used for validation of the PROWATER SPM in the 2SeasRegion

Country	Catchment	Location of validation sites	Site description	Validation method
NL	De Mark	Strijbeekse Heide (19 sites)	Nature reserve consisting of sandy soils covered with heather, pine forest and agricultural fields, often intensively drained. Upstream part of catchment.	Visual validation based on historical maps, seepage patterns, land use drainage patterns, aerial photographs and field visits. Comparison with groundwater levels.
UK	Kentish Stour catchment	Little Stour & Nailbourne catchment (4 sites)	Highly permeable chalk bedrock sometimes covered with impermeable top-layer (karst). Groundwater dominated system with deep groundwater and few streams and wetlands.	Visual validation based on satellite imagery, habitat maps, additional models (flooding and surface water runoff) and field visits. Discussions with stakeholders. *Additional monitoring is planned.*
UK	Cuckmere and Pevensey Levels catchment	Friston Forest (3 sites)	Mostly exposed chalk bedrock. Dominated by mixed deciduous forest, chalk grassland and heathland. Groundwater dominated.	Same as Little Stour & Nailbourne
UK	Medway catchment	River Beult (10 sites)	Clay catchment, impermeable topsoil exacerbated by intensive land management and loss of wet habitats. Surface water-dominated, extensive network of streams, drainage ditches and ponds.	Same as Little Stour & Nailbourne
BE	Kleine Nete	Kasterlee & Lille (2 sites)	Agricultural fields on sandy soils. Upstream part of catchment with extensive stream network	Visual validation based on aerial photographs and field visits.

6.1. The Netherlands

6.1.1. Study area

Figure 16 shows the location of the validation sites at Strijbeekse Heide within the larger catchment of De Mark, located on the border between the Netherlands and Belgium. Strijbeekse Heide is a typical infiltration zone between two small rivers the Boven Mark and the Chaamse Beken. It is a nature reserve consisting of sandy soils covered with heather, pine forest and agricultural fields (Table 2). Figure 17 shows the results of the application of the PROWATER SPM on the Strijbeekse Heide. Three types of wetlands are identified: upstream wetlands or fens/small depressions (type A), headwater wetlands (type B) and valley bottom wetlands (type C).

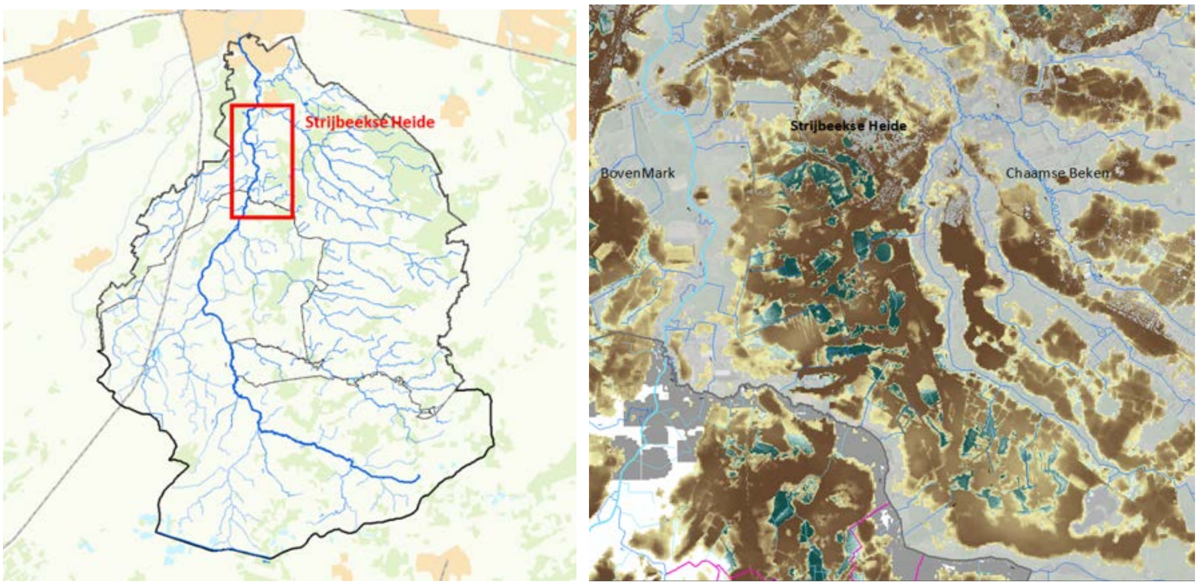


Figure 16 – Location of the validation site Strijbeekse Heide within de Mark catchment (left) and between the streams Boven Mark and Chaamse Beken (right)

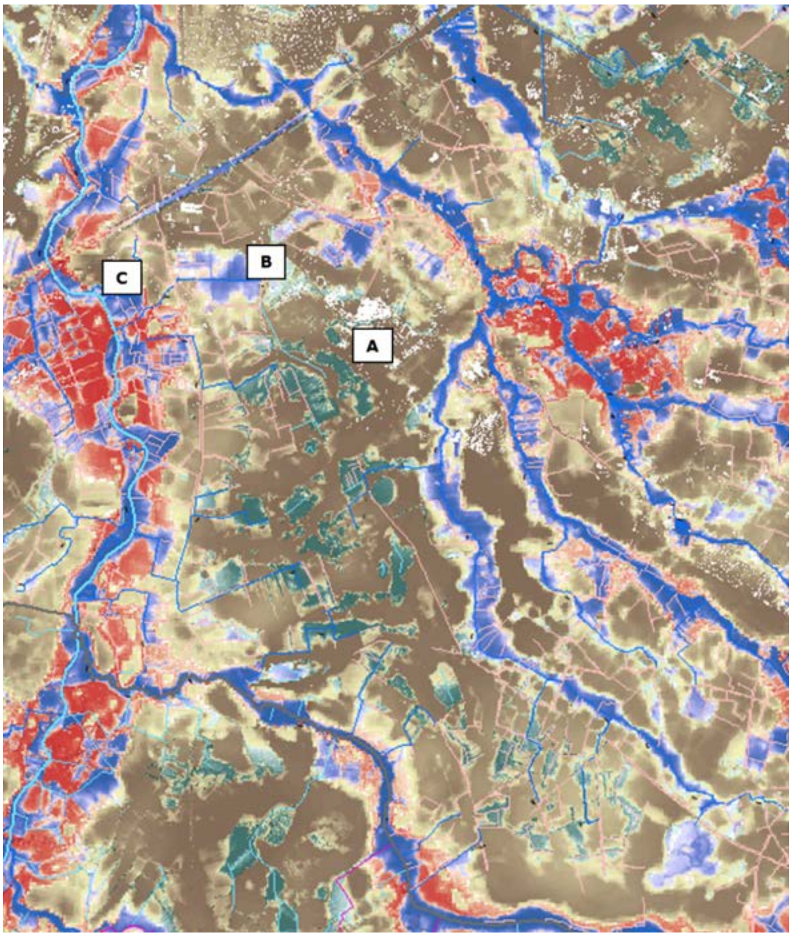


Figure 17 – Types of wetlands in the validation site within de Mark catchment (NL) as identified using the PROWATER SPM. The type A Wetlands are upstream wetlands (fens/depressions) or landscape depressions that are temporarily filled with runoff and local interflow. Wetland type B: Headwater wetlands that deliver base flow to the formation of small streams. Wetlands Type C: valley bottom wetlands with permanent groundwater seepage. Small channels are presented typical for drainage the wetlands for agricultural production.

6.1.2. Validation methodology

For 19 clusters of type A wetlands identified with the PROWATER SPM (Figure 18), a comparison was made with historical maps, seepage-infiltration patterns, land use, groundwater levels and aerial photographs or field visits. For each basis of comparison we here illustrate how the validation was done based on visual interpretation of the image or data.

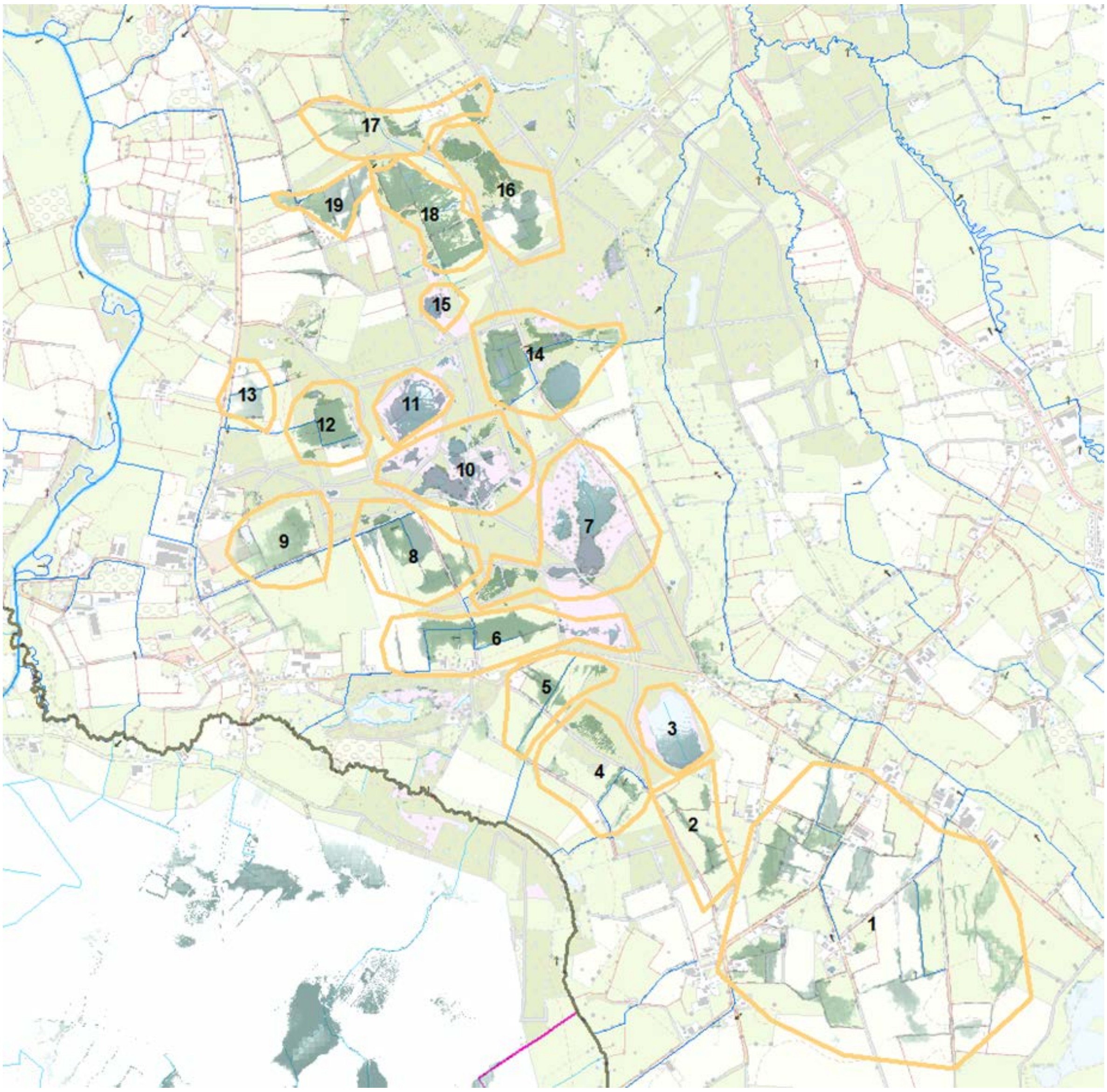


Figure 18 – Selection of 19 clusters of areas that belong to the type A wetland characterized by delayed infiltration.

In an earlier groundwater study, it has been shown that a lot of seepage water is east-west oriented and constitutes an important baseflow for the Boven Mark (Herinrichting Markdal, Hydrologische modelstudie en vooronderzoeken 2017). **Seepage-infiltration patterns** correspond well to the location and the type of wetlands identified with the PROWATER SPM (Figure 19). Type C wetlands correspond to a longer residence time of water particles in the soil, before they reach the surface or a water body (Figure 19). Type A wetlands correspond to areas with a shorter residence time.

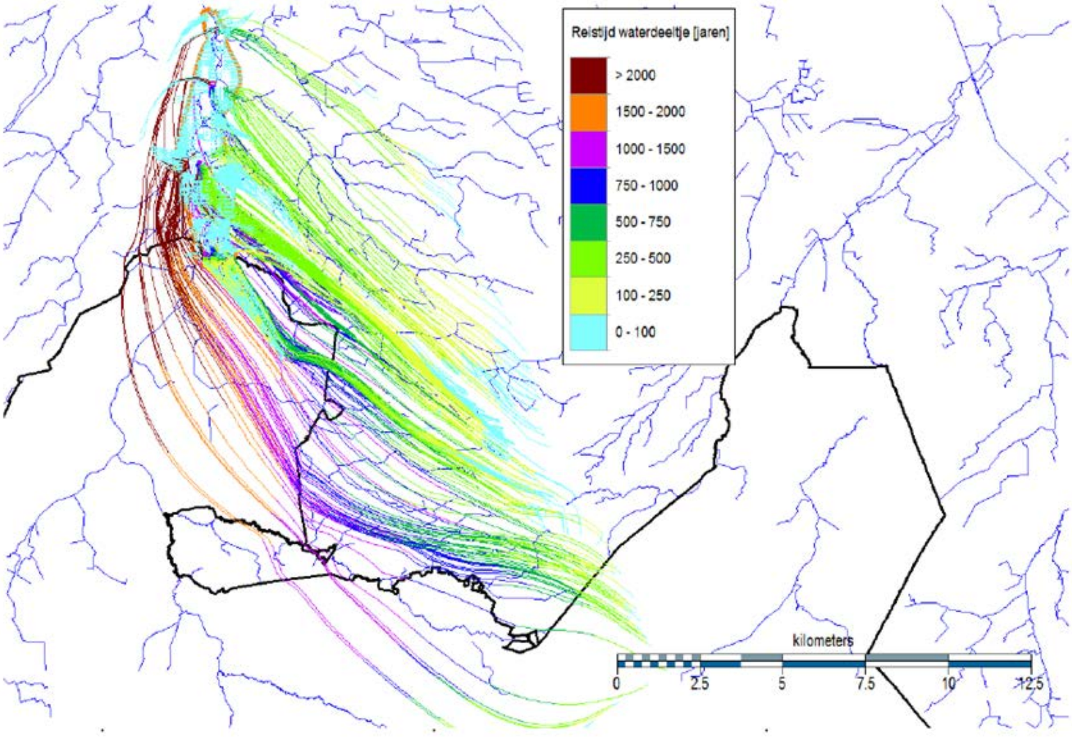


Figure 19 – Groundwater flows in the Boven Mark subcatchment. The colours indicate the residence time of the water (blue: 0-100 years- red: longer than 2000 years).

As can be seen from the overlay of the type A wetlands of figure 16b with the **historical maps** (Figure 20 and figure 21), more than half of the spots identified as wetlands by the SPM are also visible as wet depressions in the landscape dominated by heathland in 1850.

Figure 20 – Overlay of type A wetlands identified with the SPM (green zones) with historical maps of Strijbeeksheide (1850).

Figure 21 – Zoom-in on Figure . Left: historical map with the wetland depressions in the landscape. Right: historical map with an overlay of the type A wetlands (green zones)

Aerial photographs often clearly show wetlands in agricultural areas as darker spots. Some examples are shown in Figure 22 and Figure 23. With the help of the Actual Elevation Map (3D) it is often easy to see how wetland locations are drained for agriculture purposes or where 'rabatten' structures (Figure 23) are present for forestry (Figure 18, cluster 4), indicating that these zones are naturally wetland areas (Figure 24).

Figure 22 – Left: aerial photograph of wet areas on an agricultural field (cluster 16). Right: wet areas where farmer has left deep traces with heavy equipment (cluster 12). Drainage of the agricultural land has reduced groundwater levels.

Figure 23 – Up: aerial photograph showing drainage of agricultural areas. Below: Actual Elevation Map (3D) showing wetland locations which are drained for agricultural purposes or forestry (‘rabatten’) (cluster 4).

Figure 24 – Illustration of a ‘rabatten’ system used in forestry (Wikipedia, August 2019). Tress are planted on the drier ridges while channels drain excessive water from the area.

Figure 25 shows the details for (Figure 18, cluster 14). This is another example of how wetlands in close proximity to each other may strongly differ in hydrological functions due to human intervention. At the restored wetland the drainage patterns are removed/dammed. It is clear that there is almost always water on the land on the lowest parts of this parcel. It is precisely these places that are ultimately characteristic of wetland vegetation types. This comparison shows clear opportunities for ecological recovery to make these typical areas hydrologically robust and more natural (at the non-restored site).

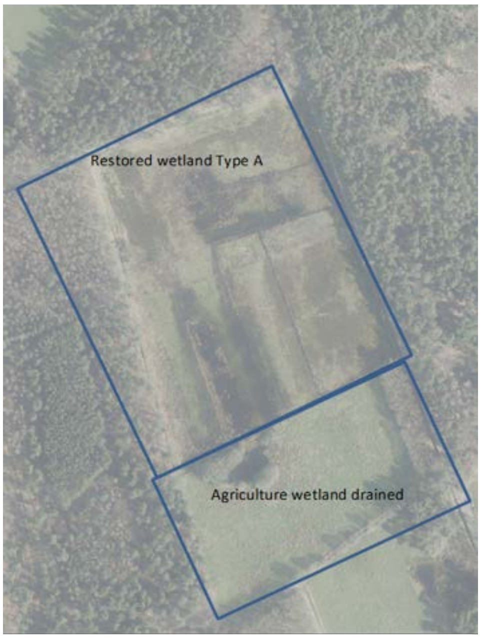


Figure 25 – Field conditions at restored and unrestored wetland (drained agricultural land)

Field visits also allow to validate physical patterns in the landscape as identified with modelling tools. Figure 26 shows the Middeltientloop, which is a typical type B wetland. It is a small seepage stream and year-round water flows from the stream into the river Mark. To restore the water levels, a typical “LOP weir” is placed close to the river Mark (figure 26, right picture).

The left picture in figure 27 shows a deep “ditch” in a production forest (figure 18, cluster 16). The right picture in figure 27 is the Rondven, a typical type B wetland depression on the Strijbeekse Heide.

Figure 28 shows a cornfield with a type A wetland. The wet depressions on the aerial photograph are clearly visible in the field. Also, the crop production is reduced on that depression sites.

Figure 26 – Middeltientloop, type B wetland

Figure 27 – Left: a deep ditch in a production forest, right: Rondven type A inundation spot.

Figure 28 - Cornfield at cluster 16 with indication of the wet area in red.

6.1.3. Results

For each of the nineteen clusters (Figure 18), it was examined whether these sites are recognizable in the landscape and on the ancillary data and whether they have the physical properties that belong to the type A wetland (table 3). Sixteen of the nineteen clusters match with wetland places on the historical map. These wetlands are mostly also recognizable on the aerial photos. 94% of all 19 wetland sites were recognized as wetland areas either on historical maps, on the seepage/infiltration map or on aerial photographs.

There are small regional differences in the physical system. On the edge of the Strijbeekseheide, seepage areas are wrongly attributed as delayed infiltration zones, such as clusters 1, 9 and 13. Ditches and dewatering patterns can be seen at almost all locations. These allow the water to drain during wet periods instead of infiltrating slowly into the soil (in natural and agricultural areas). The most promising areas that already have a nature function can create additional benefits for water retention by looking closely at the dewatering and groundwater levels. In some cases, this means that for natural purposes forest transformation is needed (removal of 'rabatten' system, i.e. drainage ditches and artificial ridges in forests).

Table 3 – Comparison of wetland occurrence using the PROWATER SPM (19 clusters) with ancillary data (historical maps, seepage-infiltration patterns, land use, aerial photographs, mean high groundwater level (GHG) and mean low groundwater level (GLG). In green: parameter is recognizable for this type A wetland.

Cluster nr.	Present on historical map	Seepage/infiltration	Land use	Present on aerial photograph	GHG	GLG
1	yes	seepage	agro	yes	30	140
2	yes	seepage/infiltration	agro	yes	70	140
3	yes	infiltration	nature	yes	-	-
4	yes	infiltration	agro	yes	50	140

Cluster nr.	Present on historical map	Seepage/infiltration	Land use	Present on aerial photograph	GHG	GLG
5	yes	infiltration	agro	yes	-	-
6	yes	seepage	agro	yes	40	40
7	yes	infiltration	nature	yes	10	80
8	no	seepage/infiltration	agro	no	70	140
9	no	seepage	agro	no	30	110
10	yes	infiltration	nature	yes	80	100
11	yes	infiltration	nature	yes	-	-
12	yes	seepage/infiltration	agro	no	50	120
13	no	seepage	agro	no	20	100
14	yes	infiltration	agro and nature	yes	30	120
15	yes	infiltration	nature	yes	-	-
16	yes	infiltration	nature and agro	yes	70	120
17	yes	seepage/infiltration	agro	yes	30	140
18	yes	infiltration	agro	yes	50	100
19	yes	seepage	agro	no	30	140

6.1.4. Conclusions

Based on this validation exercise we conclude that the results of the SPM are very accurate to identify the location of infiltration zones and Delayed Wetland types. The maps provide opportunities for further research to make the vulnerable catchment areas more physical robust in terms of changing climate conditions.

A further step that would improve results and have more accurate representation of true wetlands is to filter out of the riverbeds which are now often falsely categorized as type A wetlands by the SPM.

6.2. United Kingdom

6.2.1. Study area

First, a brief overview of the pilot areas is given, focusing in particular on landscape characteristics and geology that determine the groundwater and surface water movement in the area.

The South East of England is characterised by a complex mix of geology (Figure 29) which greatly influences not only its ecosystems and landscape but also its water resource situation. Most water for public water consumption is abstracted from groundwater aquifers. Two main aquifers are present in the PROWATER South East England region: the chalk aquifer and lower greensand aquifer. Due to their inherent geological differences – the lower greensand aquifer consists of porous sand and sandstone, while the chalk aquifer carries water mainly through fractures in the rock – they behave differently and give rise to different hydrologic regimes. Additionally, especially the chalk aquifer is influenced by overlying superficial deposits, often impermeable, which limit recharge areas and can impact the bedrock characteristics (for example by increasing the occurrence of solution features in the chalk) as well as the soil types found.

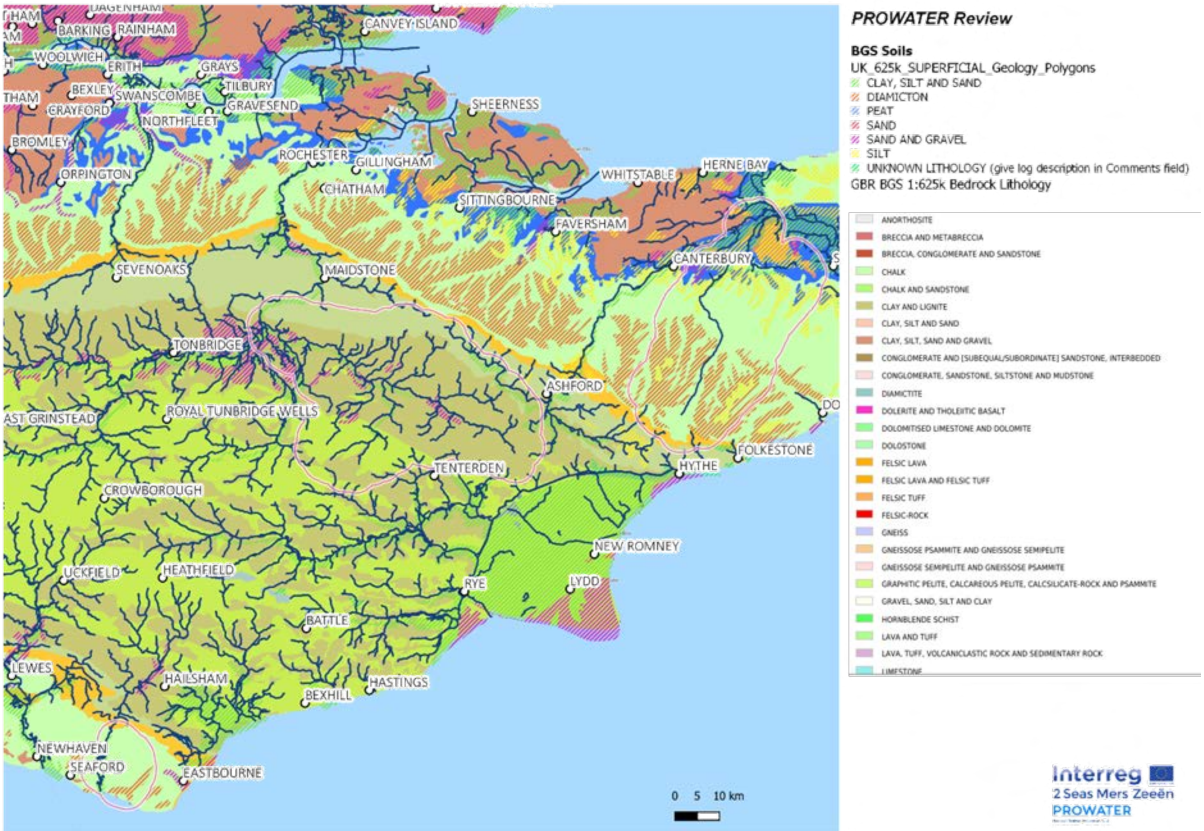


Figure 29 – Map showing the bedrock and superficial deposits found in the South East of England, as well as the main rivers. PROWATER pilot catchments are highlighted. The differences in geology as well as nature of the surface water network are easily visible.

The River Beult is one of the pilot areas in PROWATER and forms part of the Medway catchment, which drains much of the Lower Greensand aquifer (Figure 42). However, the Beult itself is mostly a clay catchment and so surface water dominated, with no groundwater in all but the southernmost part of the catchment and heavy, impermeable soils leading to rapid response to rainfall and a flashy flow regime as well as an extensive network of small streams as well as ditches and field ponds that drain the landscape.

The Little Stour and Nailbourne catchment, the second pilot area, is part of the Kentish Stour which drains the East Kent chalk aquifer (Figure 31). The highly permeable nature of the chalk bedrock and the deep groundwater level explains the lack of an extensive stream network or wetlands. Chalk rivers such as the Little Stour are mostly fed by groundwater, and it is common for their source to migrate up- and downstream depending on the groundwater level, as is the case with the upstream section of the Little Stour and the River Nailbourne. Their catchments are characterised by dry valleys and thin soils, although they are also impacted by superficial deposits on hilltops across most of the upper catchment. These superficial deposits are impermeable clay-with-flints, generating (often acidic) runoff that flows onto the chalk hillslopes and can lead to the expansion of solution features which provide rapid recharge pathways to the groundwater body, while erosion from the deposits into valley bottoms can lead to soils in valleys becoming impermeable. In addition, different layers of chalk can have different properties that influence discharge and flow of water to the surface and in the groundwater body, and fault lines or other structures can also alter behaviour. This, for example, influences upstream sections of the Nailbourne, which may appear in some years and flow for a few kilometres before disappearing again and reappearing further downstream – likely due to different chalk formations being present and a faultline influencing spring flow, complicating the understanding of which areas contribute to recharging groundwater or rivers (Aldiss *et al.*, 2004).

Lastly, Friston Forest is located almost exclusively on exposed chalk bedrock (part of the Seaford and Eastbourne Chalk Block), with only a small proportion covered by superficial deposits. No streams or rivers are found within the pilot area which is characterised by mixed deciduous forest, chalk grassland and some rare chalk heathland.


Figure 30 – Left: chalk cliffs of East Kent, South East England. Right: High Weald (Wikipedia, August 2019), part of Lower Greensand geological unit

6.2.2. Validation methodology

Evaluation of the Spatial Prioritisation Tool in the context of the PROWATER pilot areas was undertaken through a number of means:

- Desktop analysis of the output maps and comparison with satellite imagery, habitat maps, and additional models (flooding and surface water runoff)
- Discussion of outputs with stakeholders
- Area walkovers

Additional monitoring is planned for the coming months.

6.2.3. Results – Little Stour and Friston Forest

On the well-drained soils of the Little Stour and Friston Forest, little lateral movement of water through the subsurface is expected. Temporary wetlands are not a natural occurrence and streams are not fed through these networks, as detailed above. The tool outputs therefore need to be carefully considered in terms of scales at which they apply and how they can be interpreted. The macro-scale was not calculated for this catchment as it is unlikely to generate useful results, and even the meso-scale is unlikely to be able to predict patterns. The zones identified by the analysis of these overall patterns as well as microscale patterns (based on a 2m DEM) are analysed in more detail in the following paragraphs.

Within the Little Stour catchment, different soil types based on different geologies influence drainage and water retention. These different soil types will impact water movement according to topography that will be picked up by the tool outputs, but this is unlikely to reflect catchment-scale hydrologic processes. Nevertheless, these small-scale patterns could give an indication of patterns of wetness that may impact recharge potential or runoff, and so allow assumptions on improved management for resilience.

Figure 31 shows the overlay of the tool outputs, with rivers on top. It is clear that the tool picks out the river valley but also the dry valleys as seepage zones, and hilltops as infiltration zones. This does not account for the fact that groundwater does not move with the surface water patterns and along this topography, or that infiltration cannot occur on hilltops due to the impermeable deposits.

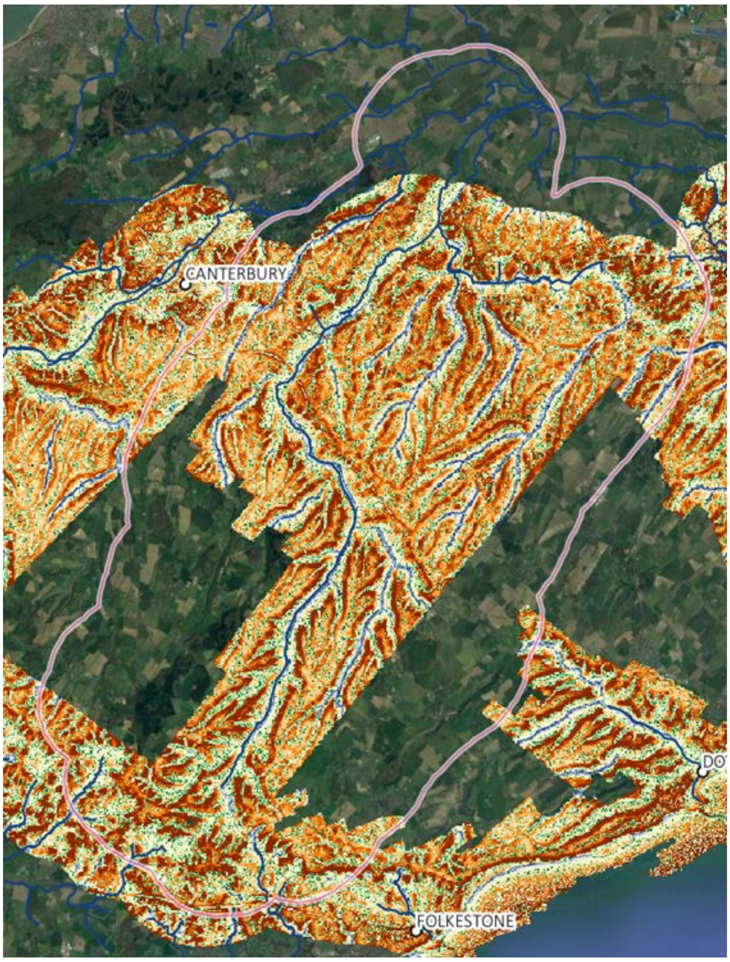


Figure 31 – Application of SPM on Little Stour catchment (UK)

Review of Infiltration-Seepage Zones

Using underlying geology and soils as a first filter, we investigate sites that are able to allow infiltration to the aquifer. This means areas directly on chalk, focusing on upstream reaches of the catchment to the South-West (with groundwater flow generally occurring in a North-East direction).

While the catchment-wide interpretation of infiltration-seepage zones does not generate an accurate picture, a focus on smaller scales allows some identification of potential patterns on the ground (Figure 32 to 34).

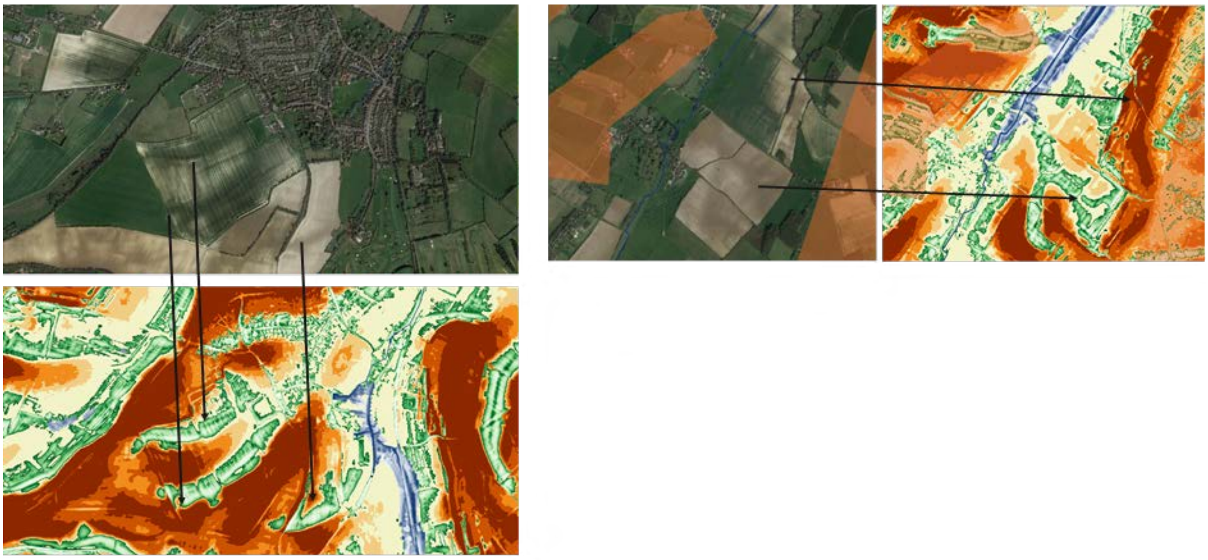


Figure 32 - Left: source of the Nailbourne showing darker patterns on cereal fields in areas picked out as temporary wetlands. Right: a few kilometres downstream from the source of the Nailbourne, also showing impermeable deposits which impact soil type and runoff patterns. Areas highlighted as infiltration zones seem drier on satellite imagery. Both areas are over chalk, with a well-drained loamy, slightly acid soil according to the Soilscapes map provided by the University of Cranfield.

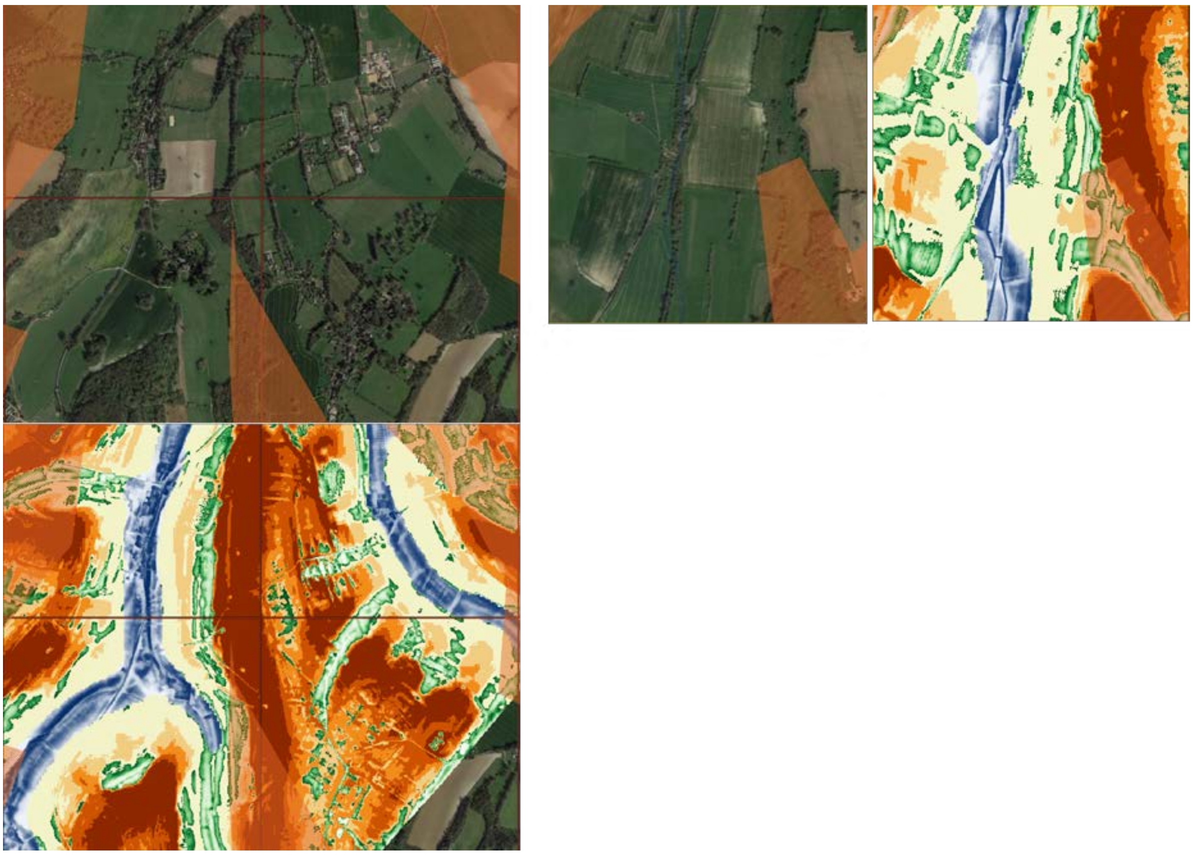


Figure 33 - Left: an area with likely high occurrence of karstic features where water reaches the aquifer rapidly. The superficial deposit is visible shaded in orange, and meets the chalk in this area, with soils mainly lime-rich chalk over the chalk bedrock. Runoff from this area can create solution channels into the rock. Patterns are less clear here than on the loamy soils shown in the more upstream areas in Figure . Above: an area of chalk soils near the main river. Fields show some differences in potential soil moisture, but it is not as clearly correlated to the patterns identified by the tool.

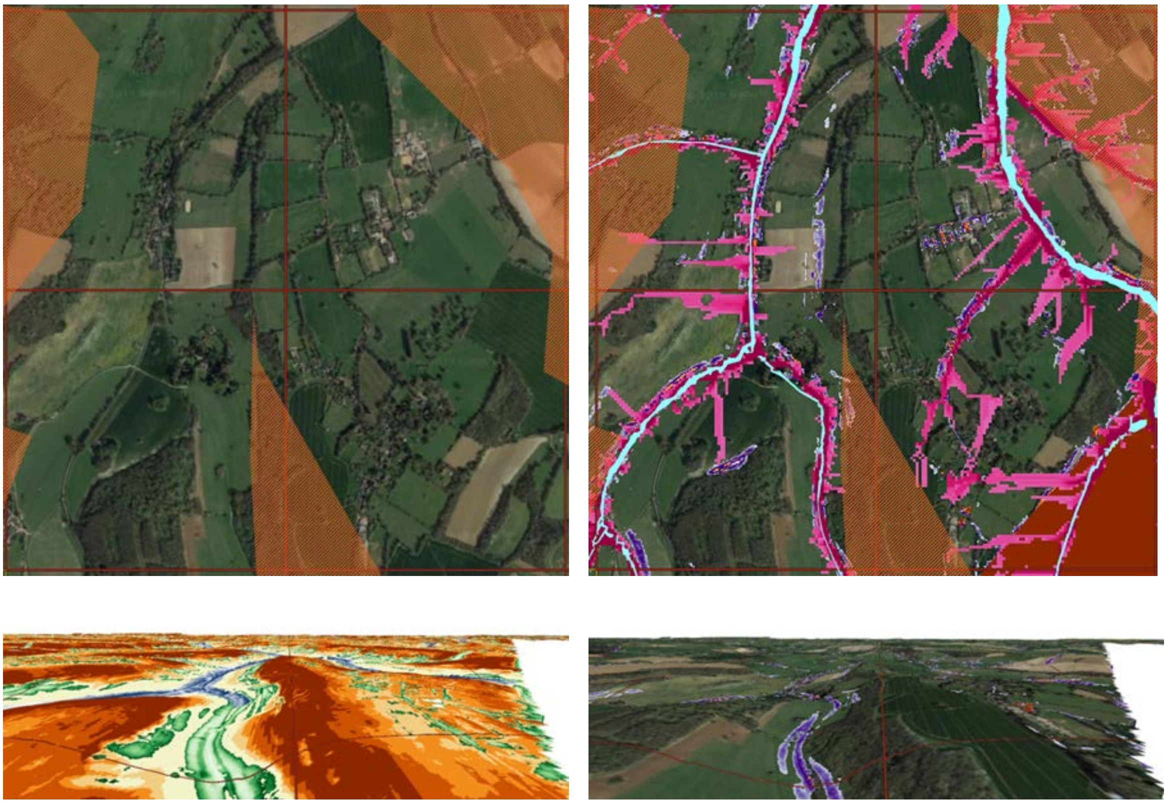


Figure 34 - Top: sites with higher likelihood of karstic features in the Little Stour show clear correlation between surface water runoff, flood risk and ditches and runoff collection zones identified by the Spatial Prioritisation Tool. These locations should be investigated further to confirm conditions on the ground and whether some of the small-scale features identified by the tool represent such karstic features. Clear runoff paths from the impermeable deposits should be a first feature to investigate. On the satellite images, it is difficult to see spatial patterns on the thin chalk soils. Bottom: 3D views of the catchment looking North downstream showing the spatial prioritisation zones and runoff attenuation areas.

Of particular interest for recharge to the aquifer are the previously mentioned karstic features. These are more likely to occur in areas where runoff from the superficial deposit (acidic) meets the chalk soils and bedrock (lime-rich) and so causes solution features. It is possible to identify these using LiDAR data, and it may be possible to use some of the tool outputs on a microscale to support this identification on a field-scale. Additionally, habitat restoration to influence evapotranspiration on particularly dry or wet areas may be targeted using the outputs above to take account of potential differences in soil moisture patterns (Deesaeng *et al.*, no date; Herbst *et al.*, 2007; Gates *et al.*, 2011).

Friston Forest presents additional opportunities to validate the outputs of the tool on a chalk bedrock with shallow chalk soils, and a variation of habitats (Figure 35). Chalk grassland and chalk heathland are rare habitats found at the site, as well as deciduous forest with some remaining coniferous plantation. Three sites are presented in this document to show the outputs in comparison to conditions on the ground (Figure 36 – 40). One site is currently farmed and the other is grazed for conservation purposes, with the last site managed for chalk heathland. No forested sites are represented as we could not find variations on the ground that allow conclusions on the tool.

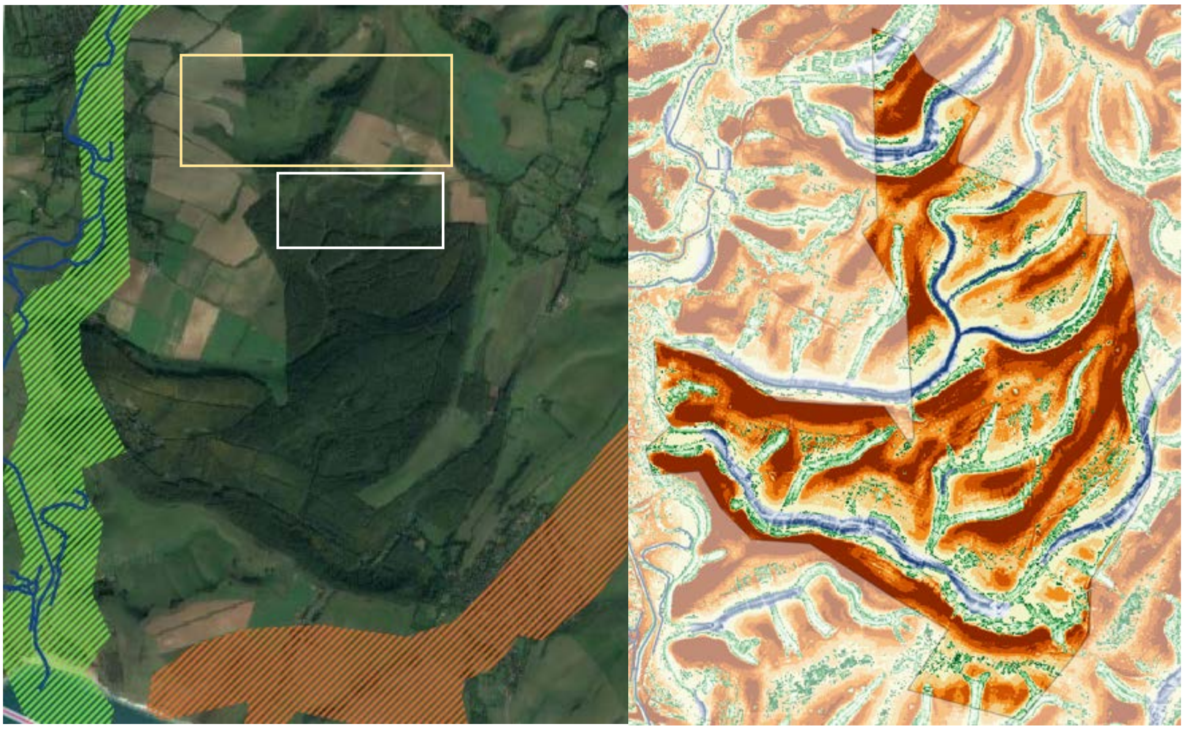


Figure 35 - Left: Overview of the Friston Forest area, with Lullington Heath (chalk heathland) visible in the North (white), and arable and chalk grassland areas (in yellow Deep Dean and adjacent areas). Shaded areas are superficial deposits on which recharge to groundwater is impeded. Right: Overview of spatial prioritisation patterns.

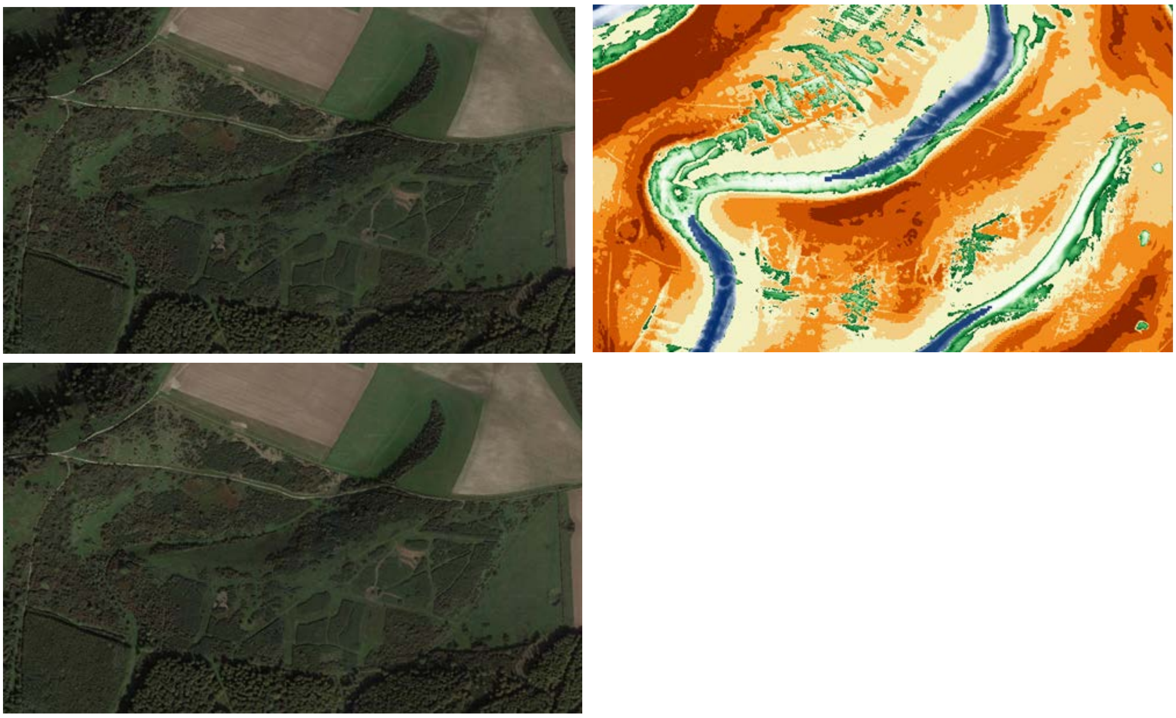


Figure 36- Left: close up satellite image of Lullington Heath area, with heathland and scrub visible. Seepage zones (in blue on the right) are potentially representing slightly darker zones on arable and grassland. It is difficult to find clear patterns of zones on the satellite imagery.

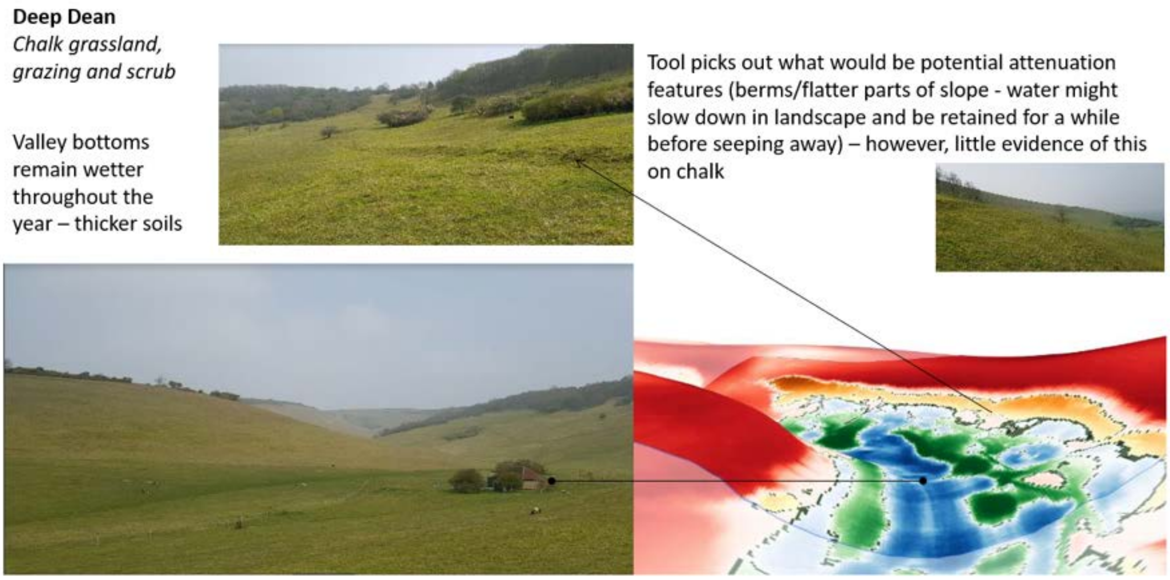


Figure 37 - Images from a site visit in the spring. The seepage zones do appear in a darker, lusher green. However, it is not clear whether this has any implication on recharge or seepage or is due to thicker soil layers in the valley retaining more water. No streams or wetlands are present in this area. However, one incident is known in which runoff from the higher areas on the arable fields close-by caused flooding in a farm. This indicates a potential way to use some of the outputs from the tool to target runoff retention areas.

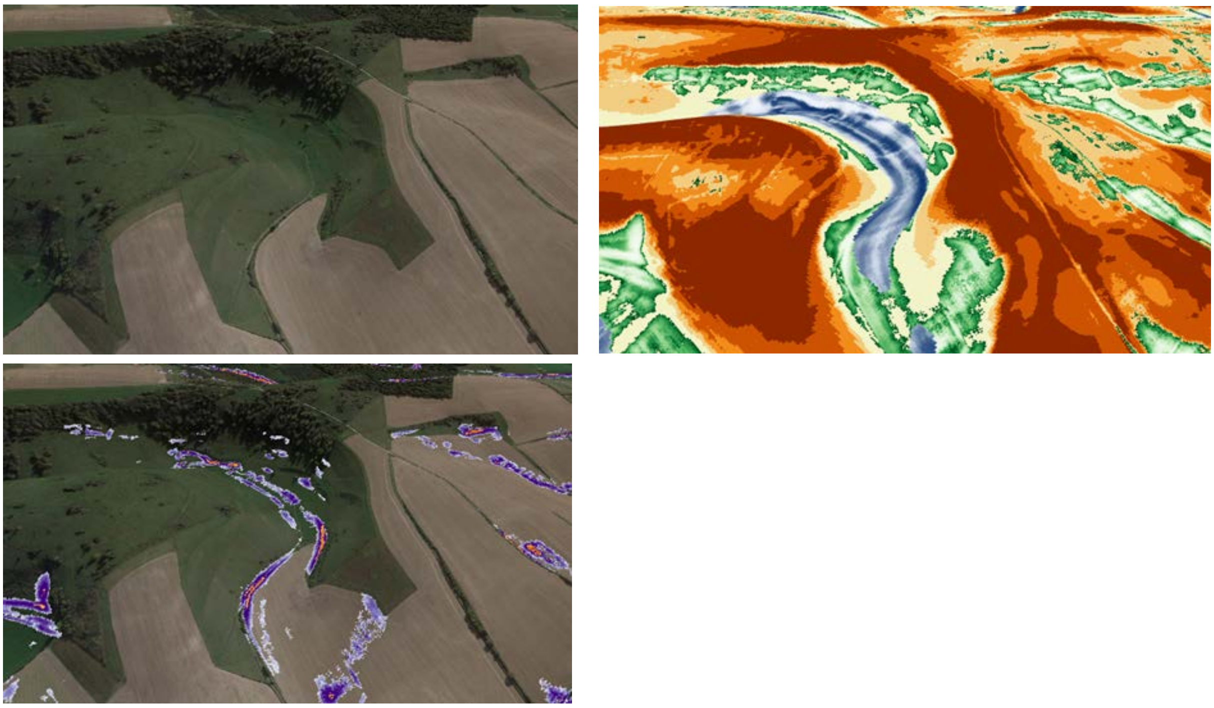


Figure 38 - 3D view (based on LiDAR 2m data created in QGIS) of the Deep Dean area in the North which is managed for chalk grassland, with some deciduous and scrub vegetation on the hilltops and slope.

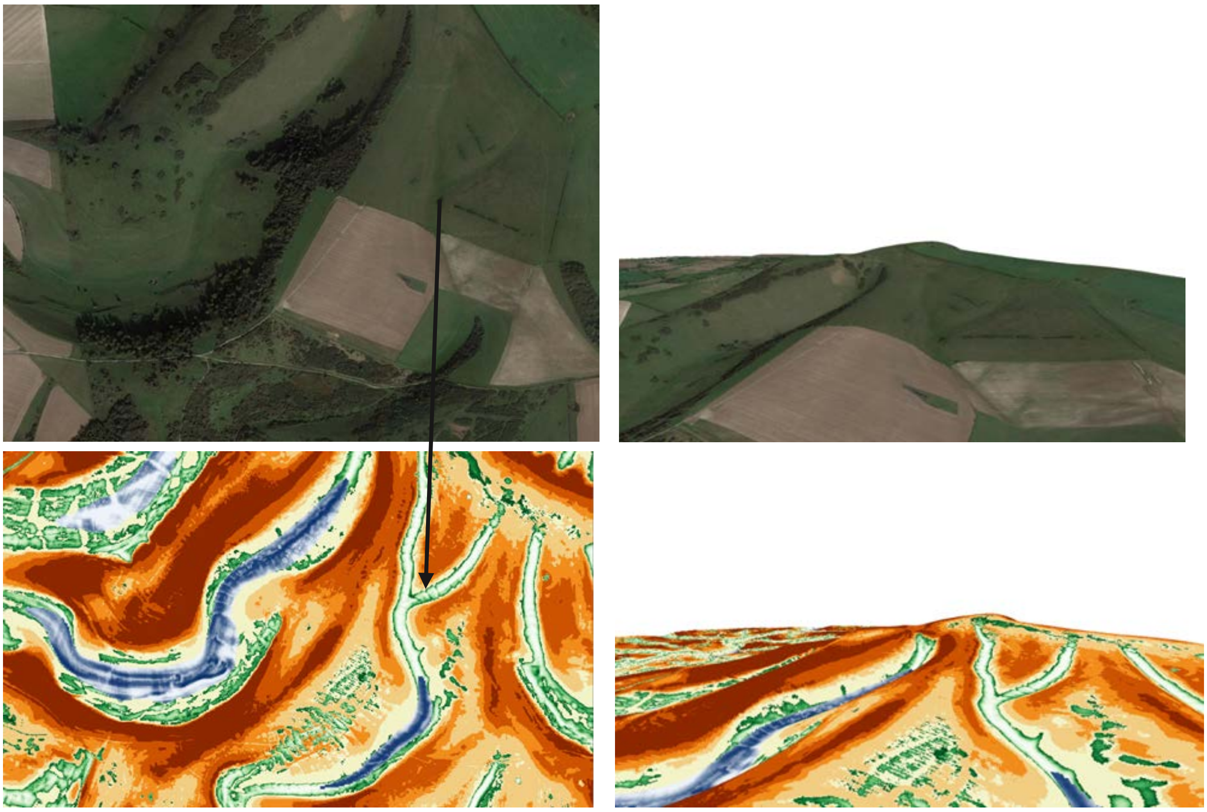


Figure 39 - Left: area in the North of Friston Forest with arable and grassland. The satellite imagery shows some patterns that reflect the spatial prioritisation zones (arrow). Right, a 3D image has been created based on Lidar data and overlain with the spatial prioritisation outputs. As these local depressions are more elevated and gentler than those presented before, it could be possible that this reflects more than just a change in soil depth and rather actually shows some small-scale water movement that could impact management recommendations for the site.

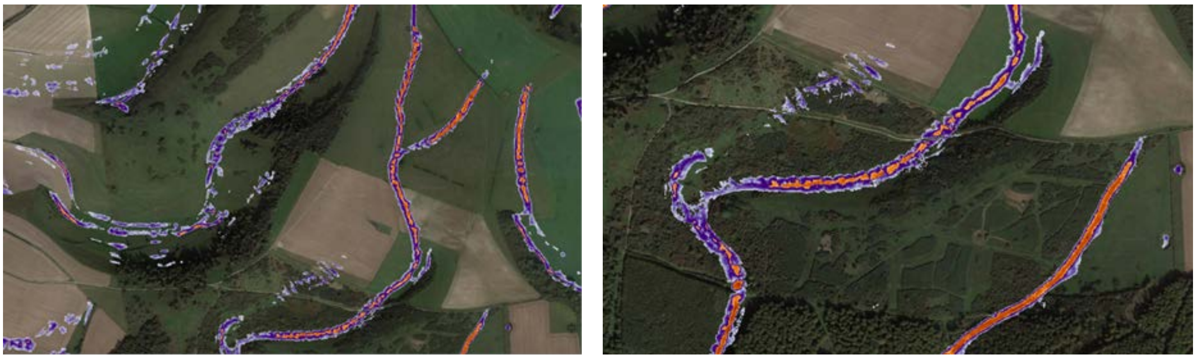


Figure 40 - Two sites in Friston Forest (left: east of Deep Dean and just north of Lullington Heath, right: Lullington Heath). Runoff attenuation areas and ditches follow clear valley lines. The darker zones visible in the left-hand image, which are identified as attenuation areas, could be on more acidic, slightly less permeable soil and so represent different seepage patterns than the thin chalk soils on most of the area.

<u>Review of potential ditches and runoff collection areas</u>

While most of the soils in the area are well drained, there are significant areas of superficial deposits generating runoff, and land use practices can increase the risk of runoff even on well drained soils, especially alongside extreme rainfall events. Retaining water and allowing it to infiltrate on the chalk rather than reaching impermeable layers or watercourses is one potential measure in these areas.

Figure 41 - In the area around the source of the Nailbourne (loamy soils), the same areas that are picked out for delayed infiltration are identified as potential runoff collection areas. On the image on the right, two additional layers are added to the map: one is a model of hydro connectivity, identifying surface water runoff pathways (purple), and the flood risk map for surface water flooding with a 0.3% chance of occurring any given year (blue). These layers confirm the potential role of the runoff collection areas for retaining water during rainfall events.

Review of micro-scale patterns

The main feature of the micro-scale pattern, the identification of potential ditches, is already reviewed as part of the previous section and will therefore not be presented again in detail. The tool is able to support the identification of potential runoff collection areas, although in this type of catchment it is unlikely that they are actual streams or wetlands. Another limitation is the lack of higher resolution DEM data, with 2m being the highest available. It is unlikely that this would be able to identify field ditches even if they were present.

Maps generated by the tool and satellite imagery were compared in a desktop assessment and discussed in meetings with stakeholders. They were presented to the internal project group as well as the catchment partnership in the Kentish Stour. Site visits were carried out in various locations on Friston Forest to understand the conditions on the ground better. This is yet to be undertaken in detail for the Little Stour.

6.2.4. Results – River Beult

The Beult catchment consists of clay soils with no groundwater, apart from areas in the floodplain in the lower catchment where local groundwater is present. The heavy soils naturally drain slowly and have high runoff rates. However, this is exacerbated by land management practices reducing infiltration and retention in the soil as well as the loss of habitats that would naturally have retained water in the landscape for longer, such as wet woodlands.

The map on the right shows the catchment outline and tool outputs alongside the main rivers. It is obvious from the extensive stream network that this is a much more surface water dominated system. While there is no actual groundwater, water moving through the soil is likely to follow the patterns predicted by the tool outputs to some extent.

Figure 42 – Catchment outline and tool outputs alongside the main rivers of the River Beult catchment

While geology varies slightly across the catchment (Figure 43), most of it is on clay with only the southern tip and northern edge on sandstone. Soils are predominantly clay soils with impeded or slow drainage. To have a first validation of tool outputs on the ground, a walkover was conducted (sites shown on Figure 43 as dots). Care was taken to visit different zones identified by the tool on broadly similar soil types and across a range of land uses.

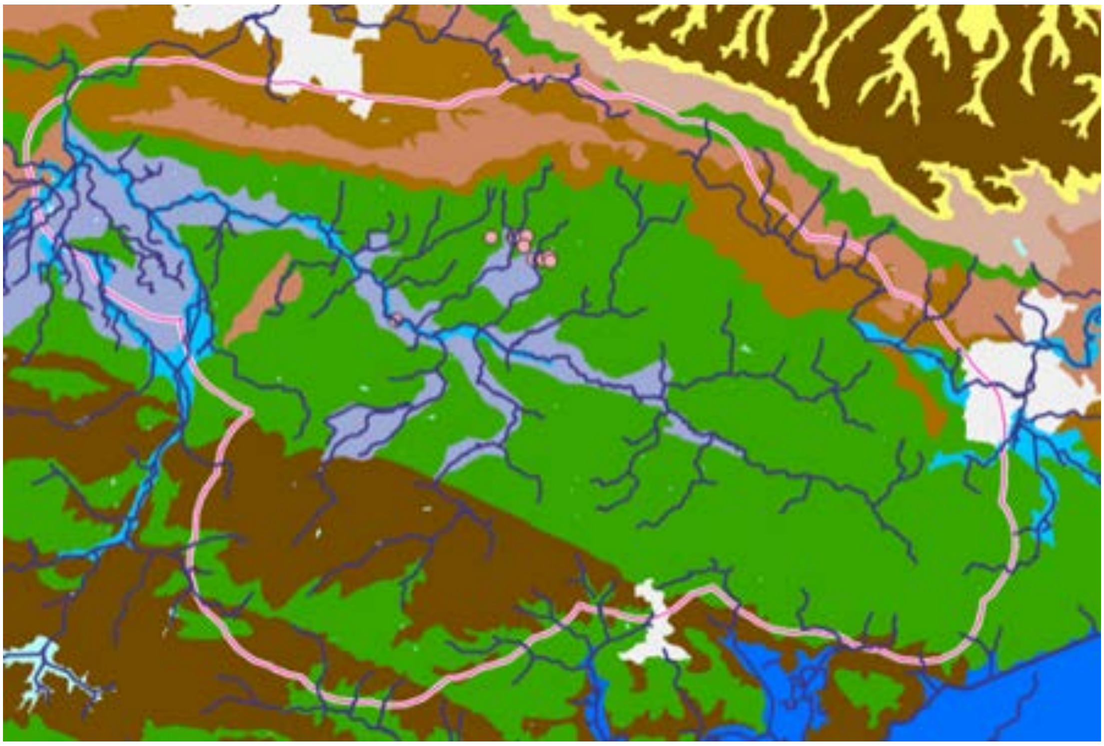


Figure 43 – Geology at River Beult catchment. Dots represent fields sites visited for validation.

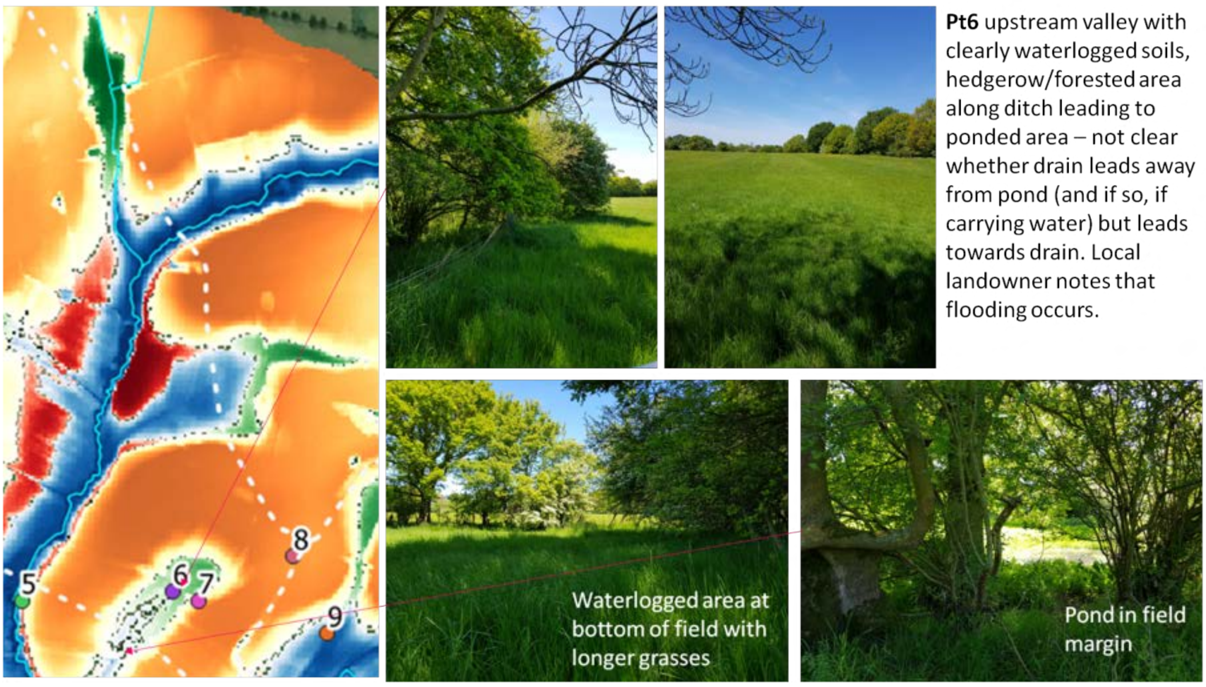
Review of infiltration-seepage zones

A first desktop review of the zones identified presented a picture more closely related to conditions on the ground than the outputs on the well-drained chalk soils. Comparing the different zones to the stream network and zones identified as floodplains indicated a reasonable alignment (Figure 44). Interestingly, some areas alongside the main river channel were shown as infiltration zones rather than the peak flow control zones that would have been expected, likely due to the incised channel conditions in large parts of the catchment. On satellite imagery (Figure 44), it is harder to identify zones easily than it is on the loamy soils of the Little Stour catchment, but the tool outputs help identify old channels where the river has been straightened.

The walkover (Figure 44), conducted in May, confirmed the ability of the tool to predict drier and wetter areas based on the calculations undertaken then, which are slightly different to the updated outputs of the tool.

The walkover included arable land as well as pastures. Pastures showed more visible differences in accordance with tool outputs, while arable land (at least where it was recently ploughed) showed little difference on the ground as well as satellite. Field drainage needs to be taken into account when interpreting the differences here, as it is not always obvious whether drains are present and still functional. Drainage ditches are very common.

Seepage and delayed infiltration zones in pastures tended to be characterised by wet or even waterlogged soils, lush vegetation and sometimes a different plant community more indicative of long-term wet conditions.



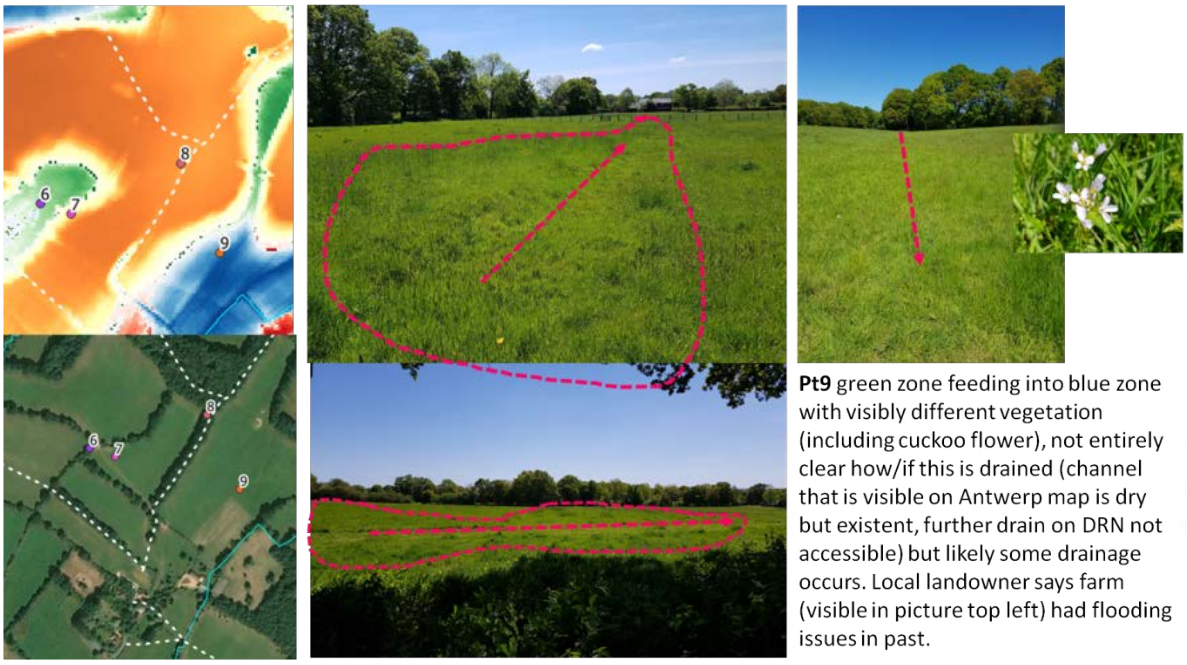


Figure 44 – Some results of visual validation of the SPM by field visits at the River Beult catchment

Arable fields at this time of the year were either ploughed or growing winter cereal. Visible differences in soil conditions were not recorded.

It is notable that in many cases ditches will begin or follow especially the delayed infiltration zones, indicating that the wet conditions there are problematic for farming. This in turn allows the conclusion that reducing this artificial drainage could support a more natural flow regime. What is not clear yet is the movement of water between the different zones. Heavy clay soils naturally generate more runoff and infiltrate less water, and it is not possible at this point to draw conclusions on how water is stored in the delayed infiltration zones and how seepage to the river occurs naturally – whether surface runoff and storage is more important here than infiltration and movement of water in the soil, and at what depth water is moving through the soils.

Review of potential ditches and runoff collection areas

Similar to the evaluation in the Little Stour, Scimap modelling was used to support the identification of surface flow paths and connectivity. Field ponds, which are a common feature in the Beult and across the High Weald, are often identified, both where they are currently "active", i.e. functioning as ponds, as well as "inactive", i.e. silted up and potentially vegetated. Both variations of these features are beneficial and able to store water as well as provide additional functions. They are sometimes connected to a drainage network and sometimes off-line. Where they are connected, modification of the drainage network (e.g. through the introduction of leaky dams) could increase storage capacity and delay peak flows; where they are not connected, they could provide an alternative water source for cattle while reducing peak flows if they are in the right location.

Review of microscale patterns

Apart from identifying ditches and runoff collection areas, the microscale patterns support the identification of historic meanders that can be used to reconnect the floodplain and naturalise the channel.

6.2.5. Conclusions

While the tool seems to be able to predict some site conditions in the **Little Stour and Friston Forest** validation sites, for example areas with higher soil moisture, it is unlikely that it accurately predicts recharge and seepage functions in a groundwater system so complex as the chalk aquifers present in the study area, where geology is a key factor influencing the behaviour of water moving through soil, subsoil, and aquifer. This limits its use for prioritisation on a catchment scale and implications for catchment resilience to droughts. Spatial patterns predicted by the tool are more like surface water processes and could not be used to allow conclusions on impacts on the groundwater by implementing interventions in a particular area. This makes it harder to interpret the likely impact and beneficiaries of measures taken.

However, in combination with other targeting methods, the tool could allow management recommendations on a field scale. For example, by predicting areas that are likely to stay wet longer on a local scale, this could be used to adjust the habitat type or management to reduce soil moisture deficits, risk of leaching, or install attenuation features that could hold runoff. Additional targeting could include other models that are able to predict groundwater movement and levels more accurately, presence of solution features and flood risk, on top of geology and soil types. Some of this data is available through the Catchment Partnerships, however detailed models are rarely (publicly) available.

In the coming months, additional validation of the tool will be carried out. On Friston Forest, discussions are ongoing about installing monitoring equipment in chalk grassland to measures differences in soil moisture profiles across the different zones identified by the spatial prioritisation tool to confirm whether the tool is able to predict long term seepage patterns on these well drained soils. In the Little Stour, walkovers will be carried out to speak to landowners and get their feedback on the outputs as well as assess the situation on the ground. Ideally, some of these walkovers will be undertaken during or immediately after rainfall events to assess runoff patterns and flow paths. Sites can then be selected to assess these patterns long term as well as establish a baseline for interventions. So far, the tool seems to predict potential wetland locations well for the validation sites at **River Beult**. A better understanding of the processes behind the patterns of wetness identified is needed to ensure interventions are designed for the right impact. This will be gained through additional monitoring over the winter months as well as site visits and conversations with landowners.

6.3. Belgium

6.3.1. Study area

Flanders is internationally labelled as a risk area for water scarcity and water shortages have already occurred in the summers of 2003, 2006, 2011, 2015, 2017 and 2018. Each of these drought episodes has had serious economic and ecologic impacts. But water supply from groundwater is at its limits. Of the 42 groundwater reservoirs, eight have too low water levels. Flanders did implement a number of measures for groundwater restoration at the request of Europe, but the Court calculated that these measures have many shortcomings (BVB, 2015). Groundwater in the Campine Region is Flanders most important buffer to overcome water shortages.

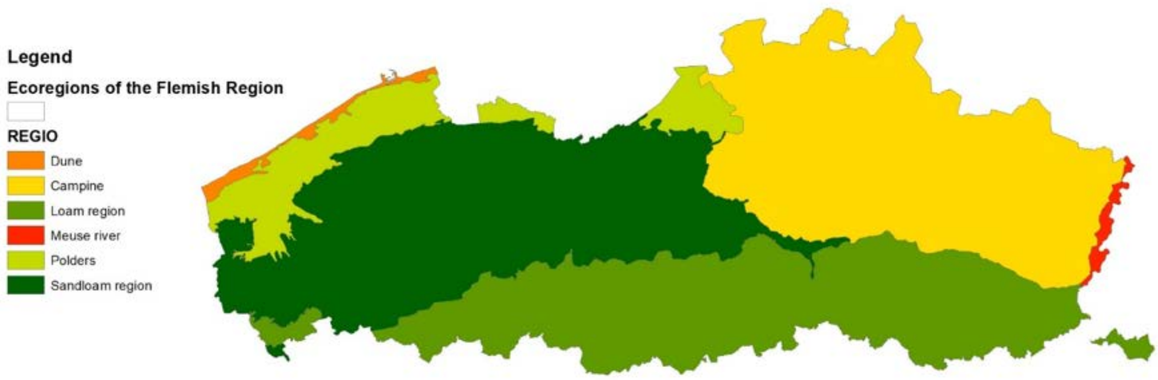


Figure 45: Map of Flanders with indication of different ecoregions: dune area, polder area, sand-loam region, Campine region and loamy region; the location of Flanders has been marked on the map of Europe.

Because of the sandy soils and the large area of forest and nature, the groundwater reservoir is large and of good quality. In the Campine Ecoregion, about 13% of the long-term annual precipitation surplus (1,340 m³) is annually extracted (175 mio m³). The Campine area is characterized by a high infiltration capacity due to the sand-cover and a relative flat surface. Furthermore, the region has a high water retention potential, which remains largely unused because of human-induced drainage. The geology comprises of a thick Boomse clay layer from the Oligocene which forms the impervious base of the aquifer system. Several sandy layers from the Miocene, Pliocene and Pleistocene are situated up there (Dams et al., 2008; VMM, 2008). Due to these favourable conditions of the basin, many groundwater extraction wells are located in the region. However, the extraction of groundwater to meet these needs is not without consequences. This extraction leads to declining phreatic groundwater levels and hence impacts nature, agriculture and forestry. About 80% of all Habitats Directive areas in Flanders are located in the Campine Ecoregion and many of them are groundwater dependent ecosystems.

Therefore, the Campine Region in northern Belgium will be the main study area, with a focus on the Kleine Nete catchment where partner PIDPA is extracting and distributing drinkwater. This study area has strategic importance for water production (65 mio m³ extracted by PIDPA for drinking water production in 2012).

6.3.2. Validation methodology

Firstly, we did a cross-validation with an existing wetland inventory. A historical and actual wetland map was developed by the Institute for Forest and Nature Research in 2016 (Figure 46). Their analysis shows that in the last 50 years almost 75% of the wetlands have disappeared in the Flemish region (Decleer et al., 2016). In the 1950s still 244,000 ha (19% of Flanders) could be considered 'wetland'. Currently only 68,000 ha (5% of Flanders) remains, implying a substantial loss of almost 75% of wetland habitats over 50-60 years' time. 37,000 ha (15%) has been urbanized; the rest was mainly lost as a consequence of intensification of agriculture and to a lesser extent also due to an increase in forest production (high loss through evapotranspiration). This wetland map is completely based on soil mapping data, whilst the PROWATER maps are based on topography. These completely independent data sets provide a good opportunity for cross-validation (Figure 47 & 48). The spatial resolution of the soil map is lower (in average 2 soil profiles per ha) and therefore we can expect that smaller wetlands (upstream depressional wetlands) are not captured by the soil-based wetland map.

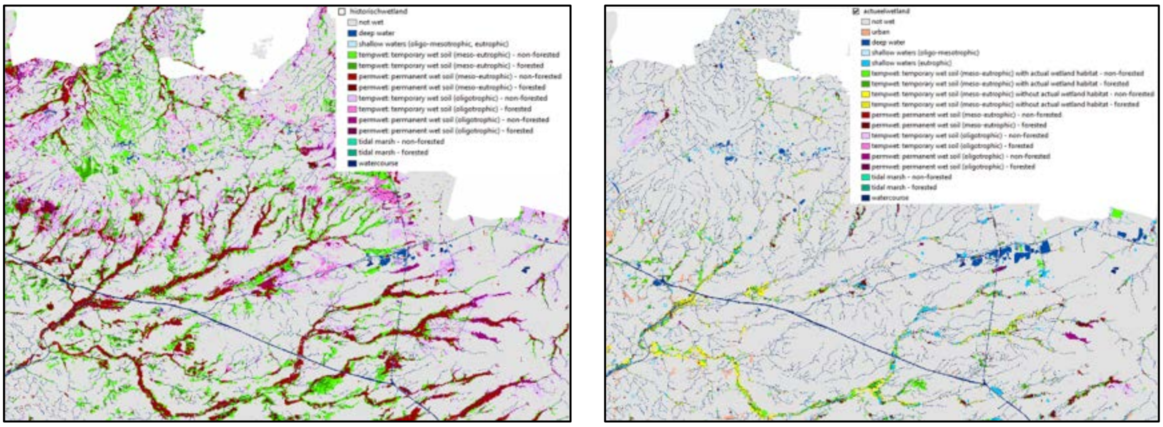


Figure 46: Historical (left) and actual (right) wetland in Flanders. Historical wetlands have been mapped using soil data. The categorisation of actual wetlands is based on vegetation data and land use data.

Secondly, we have done a visual interpretation using orthophotos (Figure 49 and Figure 51). The orthophoto series were taken on the 23th of May 2012 (AGIV, 2012). The preceding month was characterised by an abundant and frequent rainfall (104,1 mm of rain during 21 days) while the normal average is 51,3 mm during 15 days (KMI, 2018). Such episodes of high rainfall surplus generate high water levels in UDWs. These orthophoto's also display the wetter parts of well-drained croplands. This has been combined with a qualitative field survey assessment. We visited suitable locations on terrain to collect their general characteristics as drainage, vegetation and spatial extent.

Finally, we have also selected four sites that will be monitored in detail for the coming 3 years. The goal is to monitor the hydrology and nutrient fluxes of the four selected sites to gain knowledge on the system functioning of small scale UDWs. Two of these sites are well drained (under agricultural management), the two others not (under nature management). The drained sites are shown in Figure 49-51. The experimental set-up for monitoring will be comparable to those of Roth & Capel (2012) and Logsdon (2015), who also studied the water balance for (drained) topographical depressions. We purposely selected specific sites that showed optimal properties to which we could formulate a clear hypothesis on their functioning. It was for example important that the drainage system has a single outlet that can be monitored. The PROWATER wetland restoration investment sites were less suitable from a practical-technical perspective. We hope to be able to extract generic response functions to rainfall surplus (and deficit), by which we are able to assess the groundwater recharge potential of restoration scenarios for other sites.

6.3.3. Validation results – Kleine Nete catchment

Cross-validation between the DEM-based PROWATER SPM maps and the soil based historical wetland maps is here visually illustrated (Figure 47). Almost all sites that are categorised as historical wetland in the soil-based inventory are also mapped accordingly by the PROWATER SPM maps. The PROWATER SPM maps seem to capture additional, mostly temporally wet wetlands. Scattered, small scale wetlands are not captured by the soil map. The soil map based inventory is also overestimating the wet areas as can be seen in Figure 48. The PROWATER SPM is more accurate and also delivers a gradation of seepage intensity within the zones.

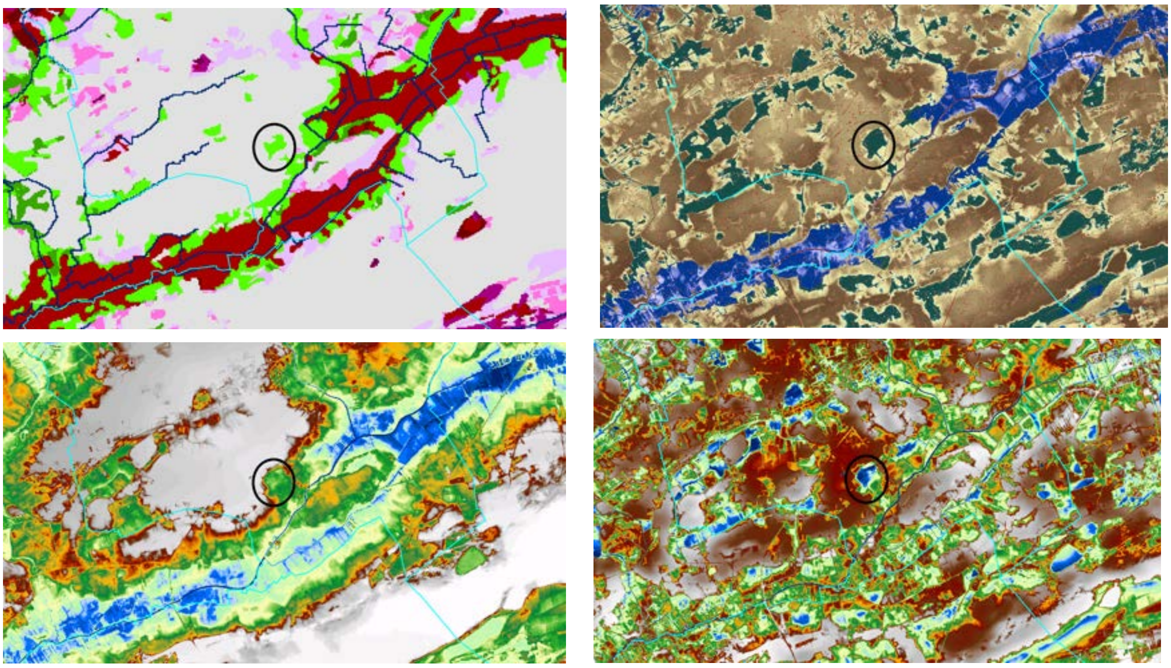


Figure 47: Illustration of the cross-validation. Top left = the soil-based inventory of historical wetlands where light green and light pink are temporal wetlands; Top right = the water system map depicting infiltration areas (brown) temporal (green) and permanent wetlands (blue). Bottom left = the meso-scale topographic position index and Bottom right = the macro-scale topographic position index.

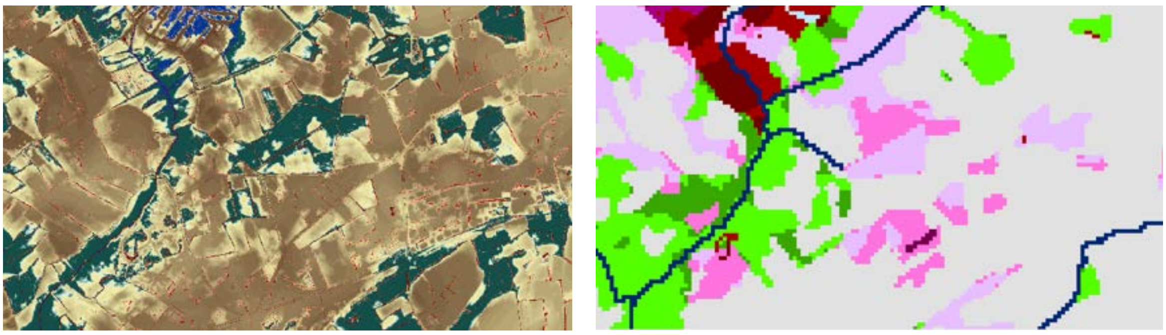


Figure 48: the soil-based wetland inventory seems to exaggerate the actual historical wetland area. This is probably due to the spatial resolution of the soil map. Mapped area corresponds to the top left corner of Figure 47.

The validation based on visual interpretation of aerial photographs and field visits is here illustrated for the sites which will be monitored in detail. The wetland that can be seen on Figure 49 is located in the Kleine Nete catchment. From the Google Maps image, it can clearly be seen that the SPM results correspond well with the wetter areas in the field (darker spots). A field visit confirmed the wetness of the area: at the lowest lying point the area is not used for corn production but is covered with nettles and grasses (Figure 50).

Figure 49 – Left: wetland type A identified with the PROWATER SPM (blue pixels, green circle) at Kasterlee (BE). Middle: aerial photograph (Google Maps) showing the wet area in the bare field (darker brown) and in the corn field (slightly darker green). Right: DHM varying between 17 (light blue) and 23 m TAW (brown).

Figure 50 – Field photograph of the site at Kasterlee (BE) showing that the lowest lying part of the area is not used for agricultural purposes (covered with nettles and grasses)

Also, in the site at Lille examples are found of good correspondence between SPM results and wet areas using aerial photographs of Google Maps (Figure 51).

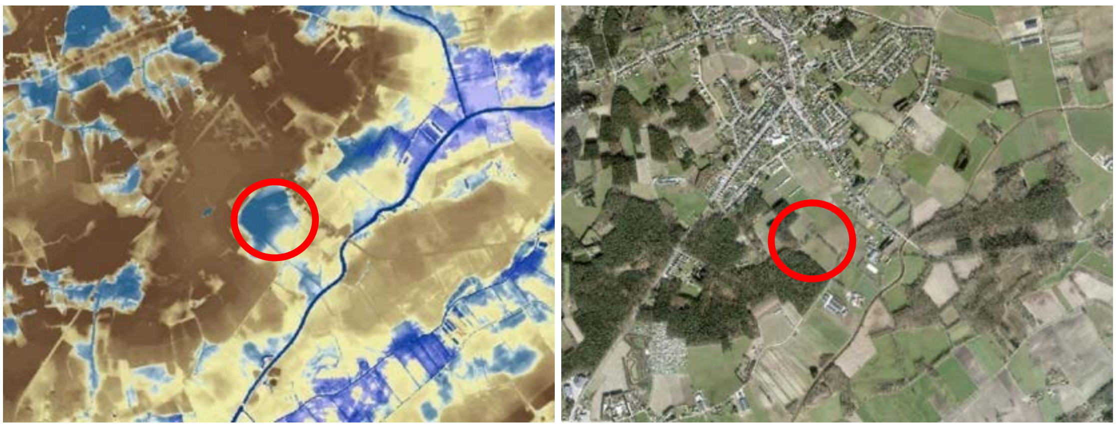


Figure 51 – Left: wetland type A identified with the PROWATER SPM (blue pixels) at Lille (BE). Middle: aerial photograph (Google Maps) showing the wet area (darker). Right: DHM varying between 11 (light blue) and 17 m TAW (brown). Study site area corresponds to the black circle in Figure 47.

7. Main conclusions

A **multi-scale TPI-based index** was developed for the mapping of priority areas for drought adaptation because of several advantages over other topographic indices. TPI-derived indices are easily interpreted, they have a relatively high accuracy (as also proven from the validation in the case-study areas) and they allow to map landforms with distinct hydrological functions. Additional pre- and post-processing techniques were applied to mimic subsurface flow and increase overall accuracy of the results.

With evolutions in LiDAR technology, **digital elevation maps** have become one of the most **widespread data** that is available on large spatial scales and at fine resolution. This opens opportunities for the development of topography-derived indices as a basis for SPM's at the catchment level. The development of several open-source tools for topographic analysis has led to a wide user-audience, including users with and without geomorphometric expertise. The literature review pointed out that the majority of SPM's based on topographic indices requires the user to make an a priori selection of the scale of the neighbourhood and that this determines to a large extent the outcomes of the calculations. The **choice of the scale** however requires insight in geomorphometry and landscape patterns which users of the tools do not always possess. Also, it is determined by the landscape features to be mapped, the type of the landscape on which the study is focused and subjectivity of the user. Hence, **generic and transferable tools are needed** that increase objectivity and reduce risks of inaccurate results due to a lack of appropriate knowledge. The PROWATER approach is based on a **multi-scale assessment** to avoid a priori selection of the scale and to minimize effects related to the lack of geomorphic expertise.

The use of a derivative of the TPI also increases correctness of the results and of the interpretation of the results because the index is quite **easily interpreted**. This is especially advantageous when the index is used by people without extensive geomorphometric background.

Topographic analyses often focus on identification of individual objectives or ecological functions or perform analyses in homogenies landscapes. Spatial prioritization of drought adaptation measures requires **mapping of areas with distinct hydrological functions** (infiltration, water retention, ...) and should allow to identify subtle differences between areas with seemingly similar conditions (e.g. upstream versus downstream wetlands). SPM's for hydrological restoration also require analyses at the catchment scale, where both **small scale and large-scale features** are of matter. A multi-scale assessment and use of **standard deviation** allow to map subtle differences between landforms at variable scales.

Due to the multi-scale assessment, the PROWATER approach may be **less suitable to be applied on very large spatial scales** (e.g. region, country) because of the extensive calculation time that would be needed for the small radii. In certain specific cases, the index as proposed in PROWATER may be insufficiently accurate and require inclusion of **additional parameters** such as subsurface features. This is for example the case when swallow holes or karstic phenomena are present that create preferential flow paths for infiltration, leading to infiltration-seepage patterns different than can be expected from topographic analyses solely.

Based on the results of the **validation of the PROWATER SPM in three countries**, we conclude that the PROWATER SPM has a relatively high accuracy in predicting seepage-infiltration patterns on different spatial scales in surface water dominated catchments (e.g. validation sites in River Beult catchment in south-east UK) and in sandy catchments with permeable soils (e.g. validation sites in the Netherlands and Belgium).

In catchments with particular geological conditions such as chalk aquifers, the SPM seems less suitable to identify seepage-infiltration patterns on the catchment scale due to occurrence of rapid recharge pathways (e.g. karstic features), impermeable topsoil or other specificities. In these areas, the basic assumption of the SPM – i.e. that subsurface flows are defined by topographic relief and presence of streams – not always holds true. In groundwater-dominated systems (e.g. in Little Stour and Friston Forest, UK) the SPM seems to be less accurate as the SPM rather represents subsurface water processes than groundwater processes. For these catchments with specific geological conditions, impermeable topsoil or groundwater dominated hydrology, the SPM is useful to derive management recommendations on smaller spatial scales.

Some steps forward to further increase accuracy of the SPM are to filter out specific features such as river beds and, in catchments with particular geological conditions or in groundwater dominated systems, to include additional parameters such as geology, soil type, groundwater movement, flood risk, …. (as also revealed by the literature review). However, data availability is often a limiting factor.

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