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Review of quantification methods for ecosystem services of Ecosystem-based Adaptation measures to drought risks

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Abstract

This review aims to identify quantification methods that allow to assess the impact of ecosystem-based adaptation (EbA) measures on ecosystem services. This review fits in the framework of the PROWATER-project, which aims to build resilience against droughts, water scarcity and extreme precipitation events through targeted EbA measures that increase the retention and infiltration capacity of the landscape. This report starts with an overview of ecosystem services and EbA measures, followed by a literature review of ecosystem services quantification methods. The ECOPLAN-SE tool, developed by the ecosystem research group of the University of Antwerp, will be used as a basis to implement modelling approaches. The final objective is effectively assessing the impact of these measures and to provide detailed and accurate information for the future implementation of these measures.

1. Introduction

Almost all landscapes in Western Europe have been adapted to support the provisioning of shelter, food, feed, fibre and fuel. They have been subject to land use changes, including soil sealing, groundwater abstraction and drainage. These pressures have had a major impact on the hydrological system, characterized through increased peak flows, declining groundwater levels and a decrease in natural water availability. These changes have had in turn an impact on river hydrology, soil nutrient retention, soil carbon sequestration and biodiversity (Figure 1).

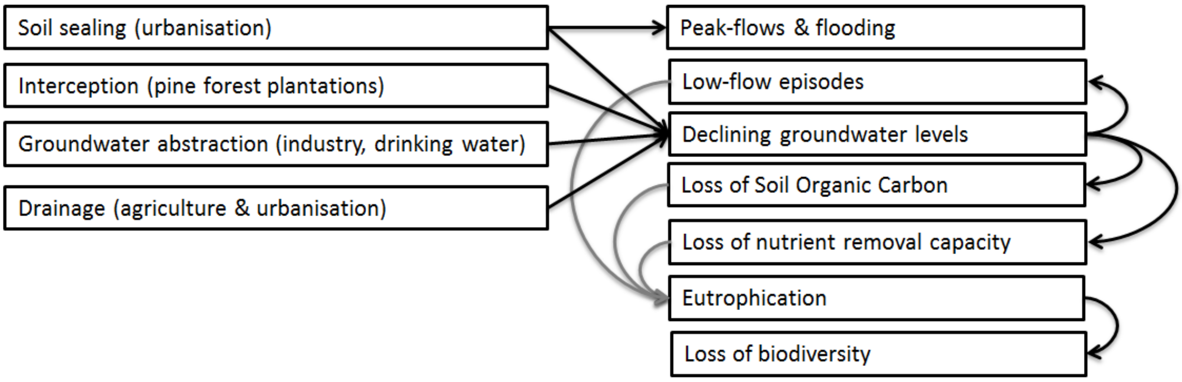

Figure 1: Pressures on the water system and related impacts

But most importantly, these changes have led to a decrease in the natural water availability and is now in combination with climate change, threatening water security will become a key. When strategic water reservoirs and/or aquifers are sufficiently replenished, drought periods and associated water demands can be bridged. However, the replenishment of these strategic water reserves has become insufficient because our landscapes have been degraded and are not adapted to deal with extreme weather.

The aim of PROWATER is to build resilience against droughts, water scarcity and extreme precipitation events. The objective of the implementation of ecosystem-based adaptation measures is to increase the retention and infiltration capacity of the landscape 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. The challenge for management and planning is to restore a natural diversity of ecosystems and create nature-based opportunities for ecosystem development that can compensate for climate changes and anthropogenic impact. This "Ecosystem-based Adaptation (EbA)" approach thus requires a new perspective for land-use planning that includes spatial objectives for the multitude of ecosystem services that need to be generated on the limited land surface.

A key objective of PROWATER is to develop modelling approaches that can assess the impact of specific EbA measures. This report reviews the quantification of water related ecosystem functions and services.

1.1. What are ecosystem services?

The linkages between nature and society are described as ecosystem services (ES), which can by defined as "the benefits people obtain from nature" (MEA 2005). The concept of ES was a central theme in the Millennium Ecosystem Assessment (MEA) report. The aim of this report was to determine the impact of ecosystem changes on human well-being and to provide a scientific basis for actions required to improve the conservation and sustainable use of ecosystems (MEA 2005). The 'Economics of Ecosystems and Biodiversity' study deals with the economic valuation of biodiversity and ecosystem services and the cost of biodiversity loss and ecosystem degradation (de Groot et al. 2010; Liekens et al. 2013). It clarifies how different economic concepts and means can help society to include nature values in decision-making at all levels.

Haines-Young and Potschin (2010) developed a framework for ecosystem services, often referred to as the "ES-cascade", where they distinguish between ecological structures and processes generated by living organisms on the one hand and the benefits that people derive from them on the other hand (Figure 1). However, it is important to mention that different authors use another definition for each of these terms (de Groot et al. 2010).

- **Structures and processes**
  Ecosystems are the functional ensemble of communities of plants, animals and micro-organisms together with the abiotic environment (Van Reeth et al. 2014). These are the ecosystem structures that vary according to the state of abiotic (such as temperature, soil type and moisture) and biological characteristics (abundance and diversity of living organisms). The difference between the two terms is that structures are rather tangible entities that can be described in quantities, whereas processes are rather reactions and operations expressed in proportions (Wallace 2007).

- **Ecosystem functions**
  Haines-Young and Potschin (2010) describe ecosystem functions to be the capacity of an ecosystem to accomplish something for the benefit of mankind, which is similar to the definitions of de Groot et al. (2010) and Willemen et al. (2008). An example of an ecosystem function is primary production. However other authors have different definitions equating functions with processes (Wallace 2007).

- **Ecosystem services**
  Many authors are using a different definition for ecosystem services. As described above, MEA (2005) describe them as "the benefits people obtain from ecosystems", similar to the definition of Wallace (2007) and Costanza et al. (1997). ES are used in different classifications. The division in provisioning services (e.g. water and food), regulating services (e.g. water and climate regulation), cultural services (e.g. recreation) and supporting services (e.g. soil formation and photosynthesis) used in the MEA-report is the most widely known. Wallace (2007) faults this classification, as supporting services are not services but processes which lead to a service. Fisher et al. (2009) describe ES as "the aspects of ecosystems utilized (actively and passively) to produce human well-being" and make a clear distinction between intermediate and final services and benefits. However, the right classification system does not exist, as it depend for which case it is used (Fisher et al. 2009).

- **Benefits and values**
  Some authors distinguish between services and benefits. Fisher & Turner (2008) define benefits as "something that has an explicit impact on changes in human welfare, like more food, better hiking, less flooding," making a distinction between intermediate and final services. For example, pollination is an intermediary service that results in the final service food, where the benefit is food for consumption. For these authors, services cannot be valued, but benefits can. The link between human well-being and ecosystems are defined as benefits, derived from services, which can be valued economically.

It is important to make a full analysis of the ES, requiring both the ecological and socio-economic aspects as well as the relationship between them to be studied (de Groot et al. 2010; Haines-Young and Potschin 2010) and therefore considering the whole cascade, whichever ES cascade is used (Boerema et al. 2016).

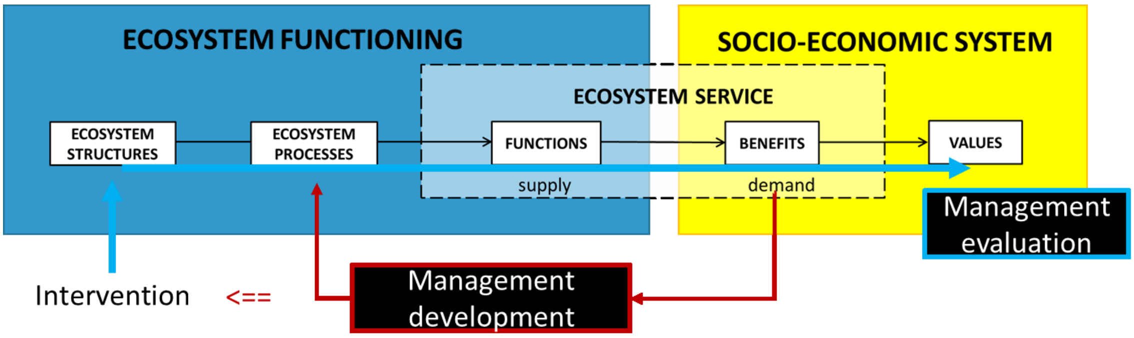

Figure 1: The ecosystem services cascade links the description of the ecosystem functioning (left blue box) with the socio-economic system (right yellow box). This forms the basis for policy development (red arrows) and policy evaluation (blue arrows). ES cascade figure from Boerema et al. (2017).

1.2. Restoring water system functioning through Ecosystem-based Adaptation measures

The objective of PROWATER is to promote and demonstrate EbA measures that aim to increase resilience against droughts and mitigate impact of extreme precipitation events. Beside their main objective, namely the increase of base flow and the recharge of aquifers and reservoirs, these measures deliver many additional benefits. For many ecosystem services, their supply is driven by (complex interactions of) ecohydrological processes (Figure 2). Therefore, a manipulation of hydrological processes will affect many ES (Brauman et al. 2007). The manipulation of hydrological pathways has been a common practice for many centuries and has only increased over time. The hydrological pathway after precipitation events are determined by evaporation (interception), runoff (soil sealing and compaction), soil water retention, infiltration and seepage (drainage). In all of these stages we can manipulate hydrological pathway to minimise losses and increase residence time.

Precipitation can occur into different forms (fog, rain, snow, …) in the watershed and may be partly intercepted by vegetation canopy. This interception loss will never reach the soil, as it will evaporate back to the atmosphere. Evaporation and transpiration (evapotranspiration) are also hydrological losses in the water balance of a system. Transpiration occurs when soil moisture is taken up by plants and evaporates from the leaves. Vegetation is also an important factor in preventing soil erosion, as roots stabilizes the soil (Brauman et al. 2007; Edwards et al. 2015; Staes et al. 2010). Soil erosion is not only influenced by topography, but also by landcover and soil management, which determine infiltration and runoff (Staes et al. 2010). By reducing runoff, soil erosion and interception, we promote infiltration and subsurface flows.

The percolation of water through the soil may also cause nutrient leaching from fertilised agricultural soils. The residence time of water is a key driver and determines the purification factor for nitrogen. Under favourable conditions, soil processes remove pollutants and nutrients before the water is deeply infiltrated in groundwater layers (Brauman et al. 2007; Staes et al. 2010). Drinking water supply is the most direct and economically most important ES delivered by groundwater. Groundwater flows to seepage zones is also important to sustain the base flow of rivers (Staes et al. 2010).

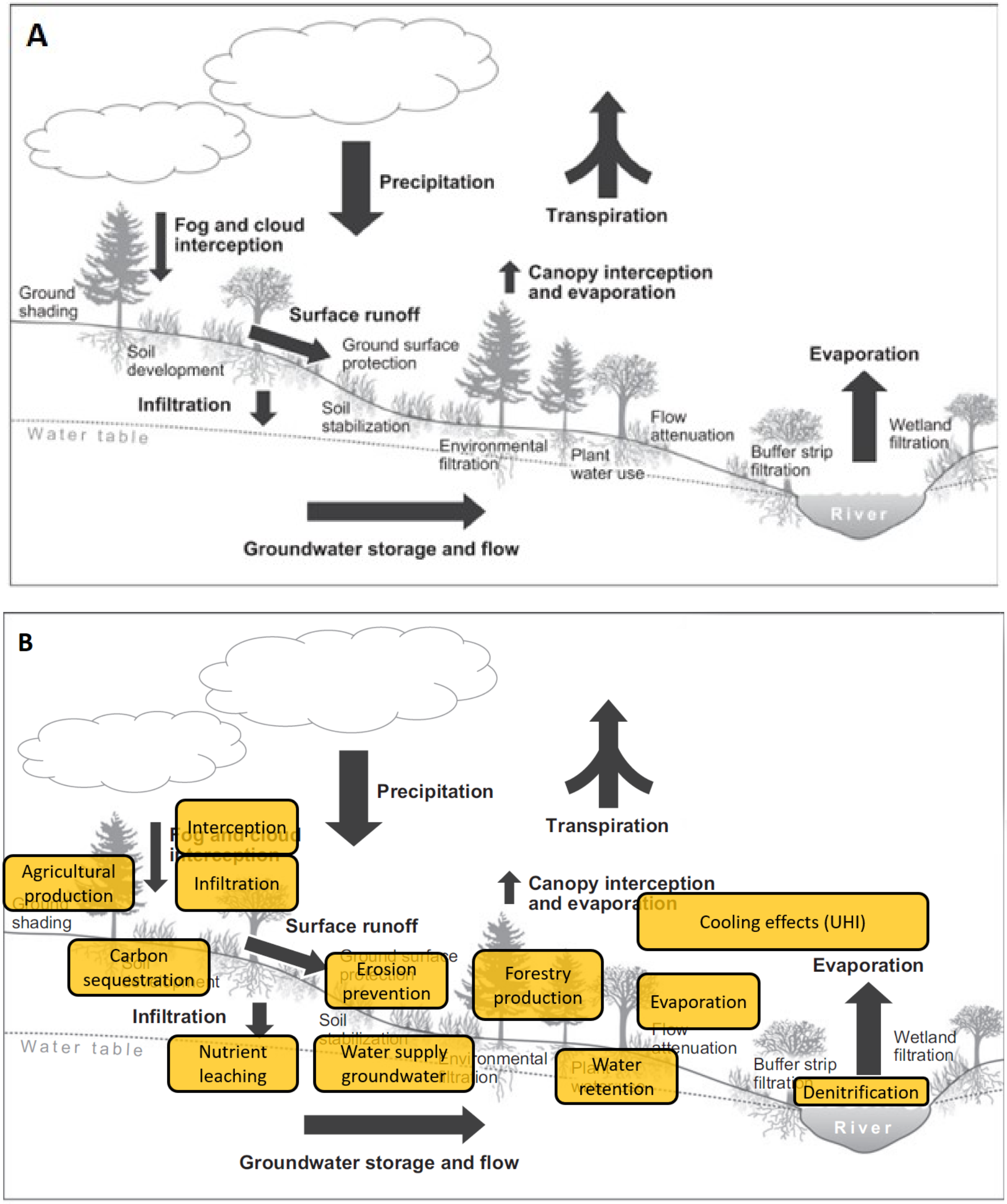

Figure 2: Illustration of the water cycle and the complex interactions with the ecosystem (a). A watershed provides additional benefits (b) (Source: own processing, based on Brauman et al. (2007))

Figure 3 is a resume of the hydrological processes and their interactions with the biophysical system. The main objective of PROWATER is to increase water stocks within the landscape and their residence time within the catchment. The residence time in influenced by diverse factors. The longer the residence time, the more water is available for water provision, which is important during drought episodes (Vrebos et al. 2014).

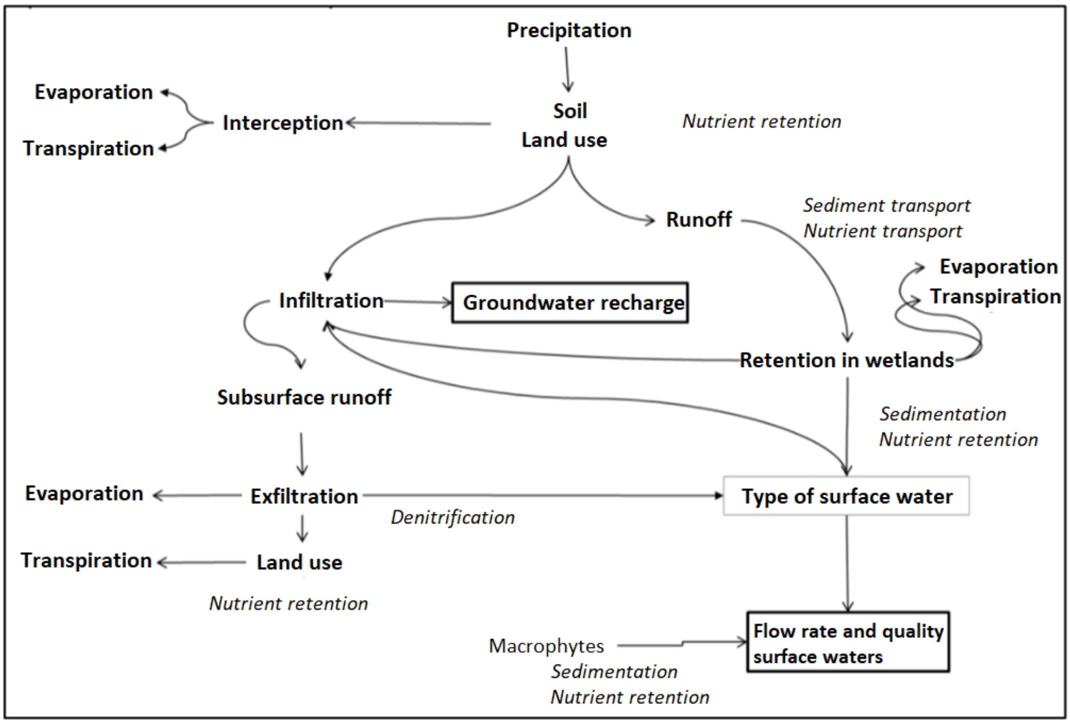

Figure 3: Schematic overview of the hydrological processes in interaction with the biophysical environment (Vrebos et al. 2014)

These hydrological interactions usually display a high spatial and temporal variability. But not all of these processes are adequately represented in conventional hydrological models. Their use has increased continuously in the last decades. They are important tools in decision making for planning and management and enable to understand the system and the quality of data records (Lain 2008). However, ES nor their underlying processes are not captured by most of these models. Most often, they have been developed and optimised for flood prediction and only capture river flow in major streams, as they are designed to predict streamflow. Models are calibrated against particular land-use and weather data. They do not allow to assess changes in land-use, and they do not capture pixel level processes throughout the catchment. Instead, they often use underlying statistical models to represent sub catchments. Advanced spatially distributed hydrological models (e.g. MODFLOW, MIKE-SHE), are better in capturing the processes at pixel level and would allow to assess scenarios. However, it can be hard to develop and calibrate these models. Other disadvantages are the limitation of the grid (extent and resolution), the low transferability and the fact that these models will not work in the case of large scale and high spatial resolution. It is also problematic that much of the current hydrology is managed or at least strongly affected by human interference. The temporal and spatial variability and heterogeneity of urbanised water systems is very different from that of a more natural environment, often resulting in much faster response of flows to precipitation surplus. It has been noticed by several authors that most models fail at the rural-urban interface (Dunn et al. 2017; Rousseau et al. 2005). Urban systems comprise of man-made infrastructure, abstraction pumps, pipes, weirs, treatment plants and so on. These elements have aspects that are in principle more easily controllable, but in practice often little is known about their exact operation and how they affect hydrology (Vrebos et al. 2014). The presence of sewage infrastructure can result in hydrological shortcuts from upstream to downstream and even between catchments (Vrebos et al. 2014). This poses a huge methodological challenge for hydrologists. Climate change and human activity are two major drivers that alter hydrological cycle processes and cause change in spatio-temporal distribution of water availability (Dey and Mishra 2017). Streamflow, the most important component of hydrological cycle undergoes variation which is expected to be influenced by climate change as well as human activities. Since these two affecting conditions are time dependent, having unequal influence, identification of the change point in natural flow regime is of utmost important to separate the individual impact of climate change and human activities on streamflow variability (Dey and Mishra 2017).

On the other hand, we do not need to model and predict the state of the hydrological system to prove the societal relevance of EbA measures. Predicting a magnitude of increase in groundwater recharge and/or avoided runoff is sufficient to demonstrate their societal relevance, especially when droughts and/or floods are regularly occurring. Within the ECOPLAN project, Staes et al. (2017) developed a conceptual hydrological model (Figure 4) based on catchment and landscape functioning theories (Blackwell and Pilgrim 2011; Jewitt 2002; Le Maitre et al. 2007). The temporal and spatial variability, interactions and feedback mechanisms are included in this conceptual model by making abstraction of temporal variability. The conceptual model incorporates basic hydrological processes and the interrelations between these processes and includes drivers and pressures. For each step in the hydrological process a high resolution spatial explicit map can be developed. Interrelations are incorporated by map calculations methods. For each process the potential performance and impact of land management can be identified, mapped and quantified. The aim is to show the potential for improvement. In a next step additional ES are linked to water and soil, for example denitrification, soil carbon sequestration and crop production.

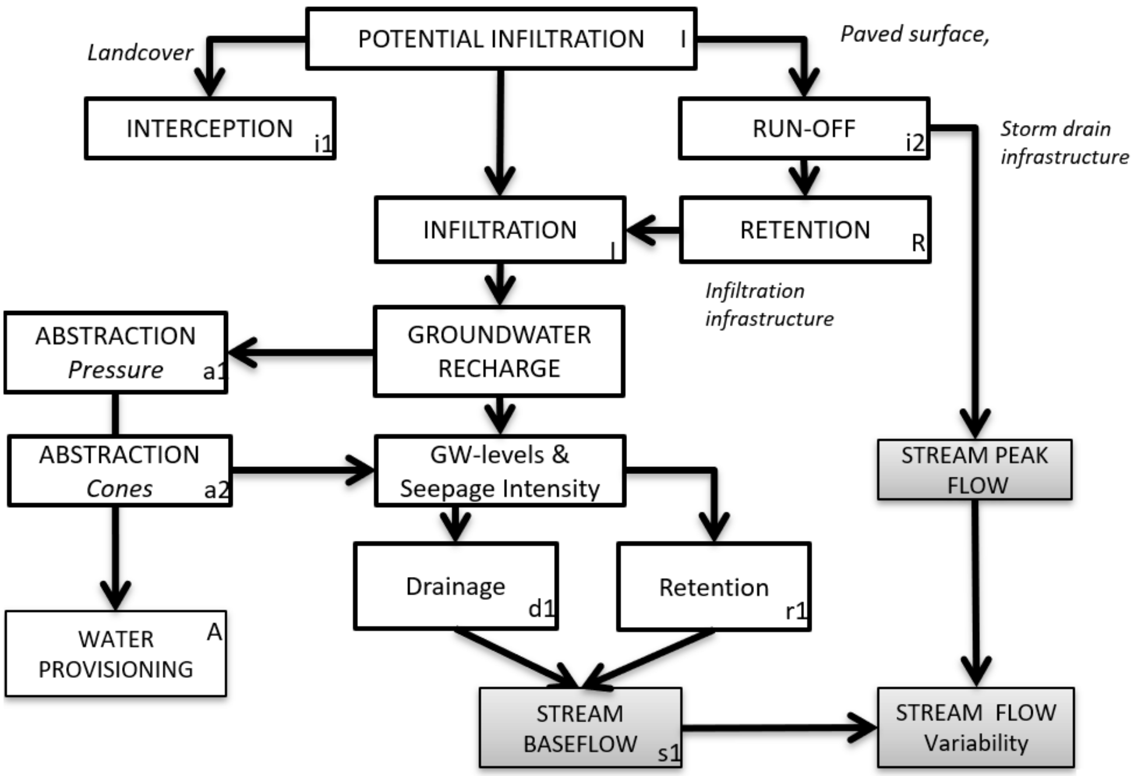

Figure 4: Conceptual model of hydrological ES (Staes et al. 2017)

The ECOPLAN-SE combined the conceptual model of hydrological ES and additional ES into one conceptual model (Figure 5). A cascade approach is used, where regulating and supporting functions are put at the top of the cascade. In contrast, most studies neglect the role of these supporting ecosystem functions throughout the modelling approach (Seppelt et al. 2011). Data with a high spatial and thematic resolution are used. The calculation methods are relatively simple and intuitive, as the basic mechanisms are captured. The output of these models is then used as input variables to model various providing services. The approach allows to incorporate shared variables, off-site effects and interdependencies that determine trade-offs and synergies between ES.

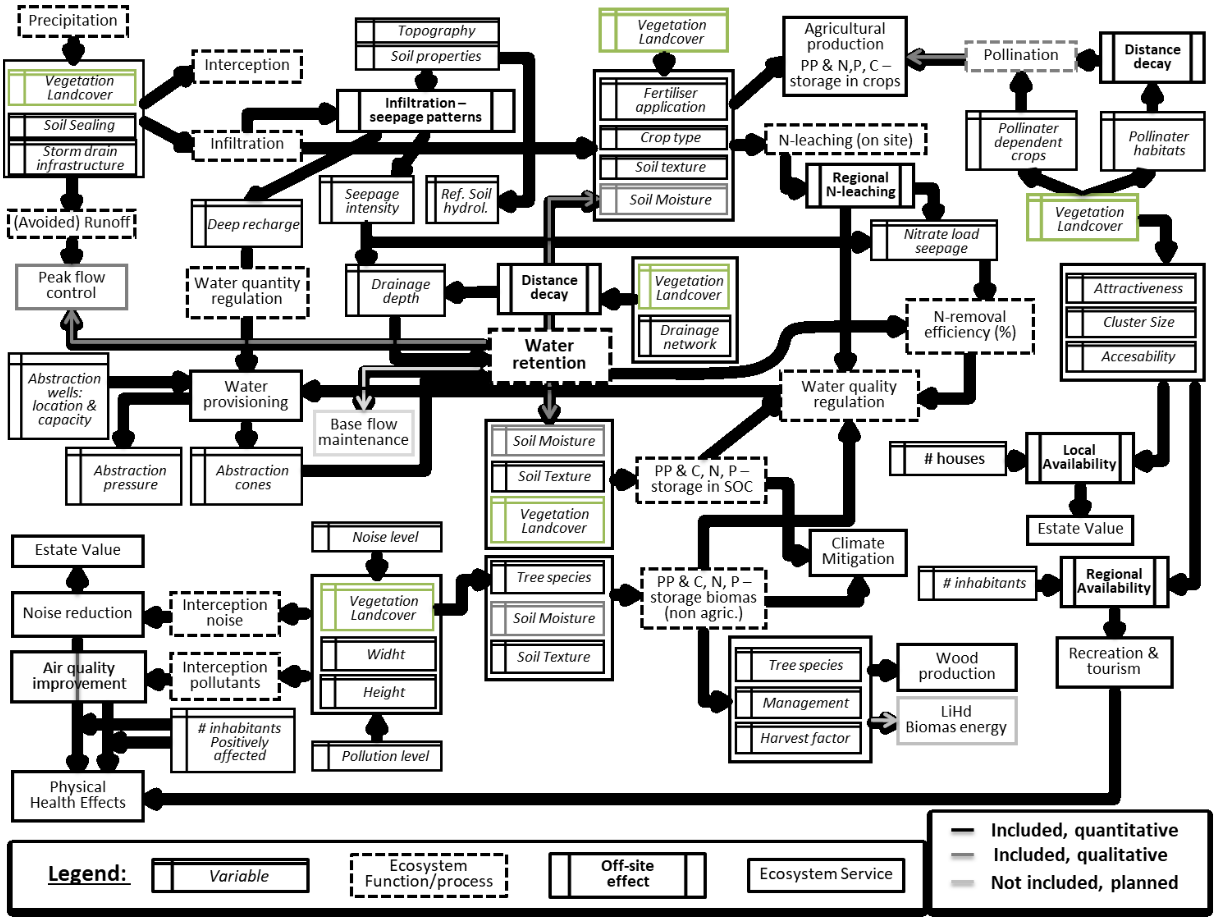

Figure 5: Schematic overview of variables, ecosystem functions (processes) and final services of the model (Staes et al. 2017)

Although the ECOPLAN-SE can quantify the impact of land-use and land-cover changes on ES, the representation of hydrological processes can be improved. This is especially necessary when we want to assess the impact of EbA measures that aim to improve infiltration and retention.

1.3. Overview of key water-related ecosystem services

This section gives an overview of the ecosystem services that are important in the context of PROWATER. The list below consists of mainly hydrological ES, but also include some additional ES that are strongly related to EbA measures.

Although water regulation is often labelled as a particular ecosystem service, it is actually an umbrella term for many different ecosystem services related to hydrology of catchments and ecosystems. Therefore, it is more appropriate to label water regulation as an ecosystem function. Ecosystem services are those aspects of ecosystem functions to which we attribute a particular societal value. Water regulation function can be approached by considering the various processes along the hydrological pathways. Promoting or decreasing particular hydrological pathway results in trade-offs. Therefore, we address the various processes that determine water regulation.

1.3.1. Interception

Interception is the amount of rainfall that is intercepted, stored and subsequently evaporated by all parts of vegetation (Cui and Jia 2014; Gerrits 2010). Vegetation and rainfall characteristics and evaporative demand determine the amount of intercepted rainfall. It plays an important role in the water balance, which is obvious especially on longer time scales. Interception leads to more gradual infiltration (Gerrits 2010).

1.3.2. Runoff

The part of the precipitation that does not does not infiltrate or evaporate will move over the land surface and through channels to end up in streams (Beven 2020; Encyclopaedia Britannica 2020). Two main types of runoff can be distinguished. Hortonian overland flow is the waterflow moving horizontally across land surface when the infiltration capacity of the soil has been exceeded (Beven 2004) and is dominant on for example cultivated soils without vegetation cover. Saturation excess overland flow occurs when the soil is saturated (Steenhuis et al. 2005).

Hydrographs are the graphical representation of streamflow over time and combines the effects of surface and subsurface flow processes in a catchment (Figure 6) (Beven 2020; Dunne and Leopold 1978). The stream discharge will increase towards a peak (peak discharge) as a response of a rainfall event. The storm runoff is finished at the end of the recession limb. The base flow is the waterflow in watercourses during rainless periods (Dunne and Leopold 1978).

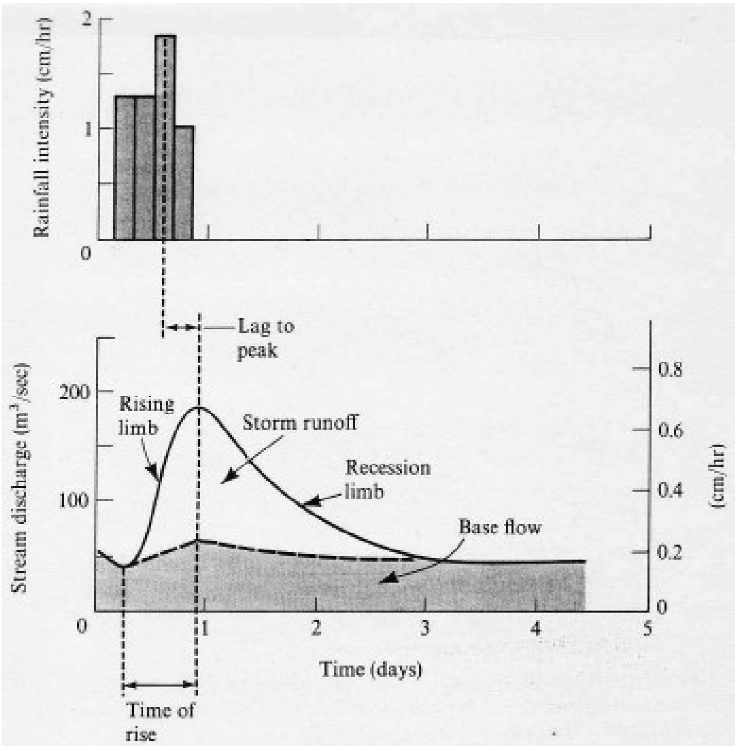

Figure 6: Illustration of a hydrograph and its components (Dunne and Leopold 1978)

1.3.3. Retention

The temporary storage of water in soils and surface waters is known as water retention. It is an important regulating service that increases the water storage and therefore reduces peak flow. Droughts are mitigated because of the sponge effect and the subsequent supply of water to groundwater and/or downstream waterbodies. Water retention supports other ecosystem services, such as denitrification, carbon sequestration in soils, nutrient retention and climate regulation. Soil characteristics, drainage and land use are determining for water retention (Staes et al. 2017, 2010).

Drainage of wetlands has been promoted on large scale since the Second World War for housing, industry and agriculture This caused decreased groundwater tables, reduced water quality, more floods and so on (Jacobs et al. 2010). The Flemish Region in Belgium has experienced an estimated loss of retention potential of approximately 13% due to drainage (Staes and Meire 2013). Retention was defined as the available stock of shallow subsurface water up to a depth of one meter below subsoil. A distinction is made between potential and actual retention, where for the latter drainage intensity is taken into account (Staes et al. 2017).

1.3.4. Infiltration

Water infiltration can be described as the movement of water from the soil surface through the soil profile (Vrebos et al. 2017). The average annual infiltration rate is influenced by rainfall intensity, soil structure, initial soil water content, vegetation and topography. Firstly, a part of the precipitation is intercepted by vegetation (evapotranspiration), another part will infiltrate and yet another part will generate runoff. Infiltration can be limited by the topsoil permeability and the subsoil storage capacity. The topsoil permeability limits the infiltration speed, expressed in mm/h, which can be relevant during extreme precipitation events. The slope gradient, vegetation roughness and soil porosity determine the ratio between infiltration and runoff for a certain precipitation intensity. For instance, soil compaction causes a decrease in water infiltration (Hamza et al. 2011; Lundekvam & Skoien 1998). The water storage capacity of the soil is influenced by soil depth and by the field capacity (the capacity of the soil to retain water against the influence of gravitational drainage (Geroy et al. 2011)). A high water table will limit the storage capacity (Vrebos et al. 2014).

Water infiltration is a regulating ES supporting many other ES, such as water provisioning (see below). The infiltration capacity has gradually decreased because of land use change and soil degradation. The Flemish Region (BE) as whole has experienced a loss of infiltration of approximately 4,3%, compared to the potential infiltration (Staes and Meire 2013). For the Nete catchment in the Campine Region infiltration losses account over 7 %.

1.3.5. Erosion prevention

Erosion can be caused by water, wind, tillage and harvesting. Whether an area is sensitive for erosion depends on topography (slope length and gradient), precipitation, soil texture, soil structure and surface roughness. Soil coverage by vegetation and litter (crop residues) can reduce the soil susceptibility to splash erosion (Van Der Biest et al. 2014).

Erosion prevention is an important regulating service encompassing the protection of the soil surface against wind and water. Taking erosion reduction measures, for example covering bare soil, results in less sediment deposition in urban area and watercourses (Vrebos et al. 2017), reduction of loss of fertile soils, reduction of sewage treatment cost and less nutrient contamination in surface water (Van der Biest et al. 2014).

We can expect more erosion due to the increasing prevalence of extreme precipitation events. Most of the erosion occurs during short bursts of rainfall (Li and Fang 2016). Figure 7 shows the different mechanisms of rainfall and temperature due to climate change on soil erosion. Despite the variable and complex response of soil erosion to climate change, future climate change will probably cause negative impacts on soil erosion, especially in regions where soil erosion is already severe. In order to maintain future soil erosion rates at no more than today's rates, or even to reduce within desirable limits, protection measures should be required. Measures that can protect soil effectively may include afforestation, conservation tillage, no-till, and planting drought-resistant cultivars (Li and Fang 2016).

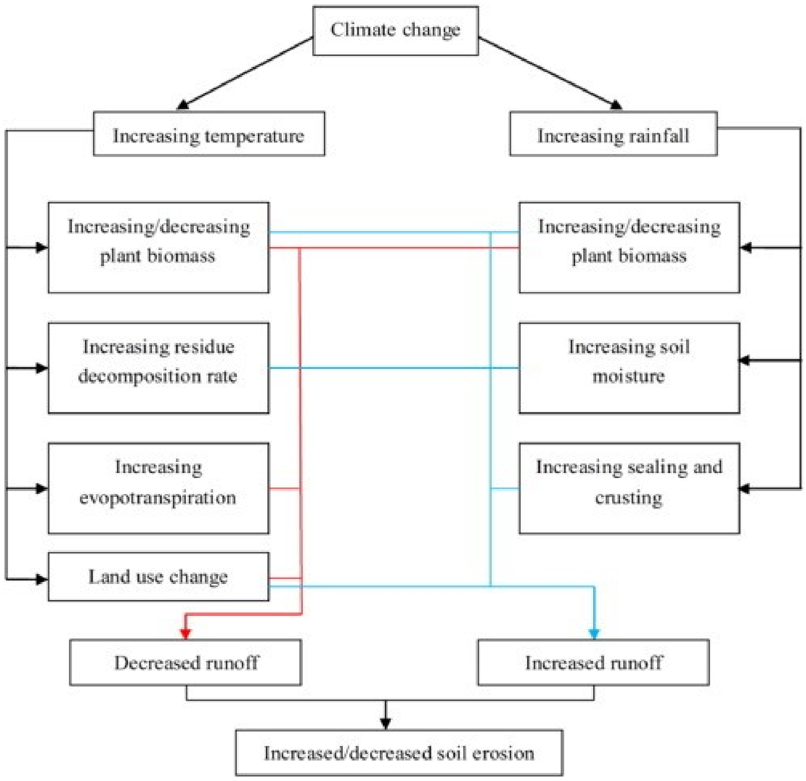

Figure 7: Illustration of the mechanisms of the impact of climate change on soil erosion (Li and Fang 2016)

1.3.6. Water purification (nitrogen removal)

Ammonium in the soil is converted to nitrate by the process of nitrification. Nitrate is in turn converted to gaseous nitrogen by denitrification. This way nitrogen is removed from to soil into the atmosphere. Denitrification takes place in oxygen deficient environment and depends on temperature, soil moisture and soil organic carbon (Pinay et al. 2007; Seitzinger et al. 2006). Ecosystems are able to remove nutrients from surface and groundwater through denitrification, thus contribute to a better water quality and is therefore a natural process of water purification. Soils with a high C/N content leach less nitrate in surface water and groundwater. Denitrification is a process that is strongly influenced by the ecosystem function of water retention (Staes et al. 2017, 2010). Therefore rewetting of formerly drained land will increase carbon sequestration, nutrient storage in soils and nutrient removal (Staes et al. 2017).

1.3.7. Climate regulation

Terrestrial ecosystems contribute to climate regulation through the sequestration of carbon and emissions of other greenhouse gases (GHG). In this part we focus on the biogeochemical processes of carbon (C), methane (CH_4) and nitrous oxide (N_2O). In general, GHG fluxes by soils are affected by soil moisture, soil temperature, soil microbial activity and vegetation (Smith et al. 2013).

Carbon

Carbon is stored in soils through photosynthesis whereby organisms (plants and algae) take out CO_2 out of the atmosphere by means of sunlight. The more diverse the species composition, the higher this primary productivity and the more resistance against diseases and pests, temperature increases and droughts. Mineralisation takes place whereby micro-organisms convert the litter in turn to CO_2 and nutrients (such as N and P) are released. Macro-organisms such as earthworms, woodlouse and mites are responsible for bioturbation, distribution and fragmentation of the organic matter. Other non-mineralized C from dead plant material is stored in the soil through humification which forms the organic matter in the soil. The biological processes of mineralisation and humification depend on temperature, moisture content, oxygen delivery, soil texture, soil drainage and so on. Another part of the C stock in the soil is derived from management operations (fertilisation). Carbon leaves the soil due to respiration, leaching out of dissolved organic carbon (DOC) and erosion of the organic material in the soil. This equilibrium is affected by changes in climatic conditions, such as temperature and moisture, soil type and land cover. Humans also play a role due to changes in groundwater level, land use and soil cultivation (Lettens et al. 2014). The C content is higher in wet soils than in dry soils because decomposition rates are slower due to lack of oxygen. Draining soils causes oxidation of C and therefore a reduction of the C content. Because of this, rewetting of formerly drained land will have a positive effect on carbon sequestration in soils (Lettens et al. 2014; Staes et al. 2017). This regulating ES is crucial for climate change, as the more C is sequestrated in the soil, the less C the atmosphere is containing (Lettens et al. 2014; Vrebos et al. 2017).

Methane

Natural sources of methane in our region include anaerobic decomposition in wetlands and emissions from livestock (Heilig 1994). Methane emissions in wetlands are a part of the carbon cycle and are strongly affected by environmental factors, such as (soil) temperature, climate, oxygen, water level and irradiance (Christensen et al. 2003; Ortiz-Llorente and Alvarez-Cobelas 2012). They are the most important source of atmospheric methane (Heilig 1994). One should take this into account when restoring water tables in organic soils (Smith et al. 2011). Methane makes up only two percent of the total emissions in our region, but is one third stronger than carbon dioxide as greenhouse gas (Lettens et al. 2014).

Nitrous oxide

Nitrous oxide gas is another GHG that is regulated by the biosphere. Emissions of N_2O are induced by microbial processes aerobic nitrification and anaerobic denitrification (Lettens et al. 2014). Major cause of this type of emissions are the application of nitrogenous fertilizer and ploughing crop residues, atmospheric deposition of ammonia and nitrogen oxides to a lesser extent (Smith et al. 2011). According to some studies (e.g. Smith et al. 2011) N_2O makes up about 3-9% of the emissions for agricultural regions. Nitrous Oxide is a potent GHG as it is 298 times stronger than CO_2 (Lettens et al. 2014).

1.3.8. Nutrient storage in soils

Nutrients are (mineral) chemical elements of which the most important are nitrogen (N), phosphorus (P) and potassium (K). Their presence is essential for growth of organisms and for the maintenance of soil fertility. Nutrients are made available for plants through mineralisation of organic matter. Input of N comes from atmospheric deposition and some vegetation types that are able to fixate N from the air (Staes et al. 2017). The nutrient storage depends on the chemical, physical and biological capacity of the soil. Chemical properties, such as acidity and cation exchange capacity determine the chemical soil fertility. The soil structure is also determining for the plants to take up nutrients. The last crucial aspect are soil organisms which are responsible for decomposition of organic matter. Humans have increased nutrient supply through fertilization, liming, crops residues and deposition from industry. This large supply has led to nutrient loss to groundwater (leaching), rivers (runoff and erosion) and to the atmosphere (Cools and Van Gossum 2014). Land use change and drainage result in an increase of mineralization and additional supply of N and P to the environment (Staes et al. 2017).

1.3.9. Water provisioning

Water provisioning is deliberately discussed at the end of the list of water related ecosystem services. It is a provisioning ES because water is taken from the natural environment to be used for drinking water, industrial production, irrigation and navigation. The extraction itself can hardly be seen as an ES. It should actually be the amount of water that can be sustainably extracted from the ecosystem. The more water available, the higher the sustainable extraction. But climate change is burning both end of this candle. While climate change is decreasing water availability, the societal demand for water grows when droughts occur. The sustainable abstraction thus becomes dependent on the amount of groundwater recharge, the filling of reservoirs and river base flow.

Water provisioning from groundwater can be derived from infiltration as a sustainable extractable amount of groundwater recharge (Staes et al. 2017). Actual delivery of water provisioning can only exist when there is an active extraction of water for use in anthropogenic activities (drinking water, industrial use, irrigation, cooling). Abstraction may be at the expense of other ES when water levels or flows drop significantly. The water abstraction pressure is thus an important indicator.

For catchments that rely on surface water reservoirs, a water yield approach is often used to assess the ES of water provisioning. The most commonly used assessment method are water balance models that show how much water is left for (base)flow and thus abstraction. But simple approaches such as the InVEST water yield model, which basically calculates annual rainfall minus annual evapotranspiration, does not allow to assess the challenge of dealing with precipitation variability. It requires input data such as root restricting layer depth (mm), plant available water content (AWC, as a proportion), average annual precipitation (mm), average annual potential evapotranspiration (PET, mm) and land use/land cover (LULC). It is evident that the model relies heavily on the quality and resolution of the input data and that the impact of measures needs to be represented in the input data layers and not within the model.

But within the context of PROWATER, the temporal dimension of water availability is especially important. For reservoirs it is important that the reservoir receives a more steady base flow with low suspended matter, rather than peak flows loaded with sediments and pollutants. Lag-time between precipitation events and river flow response is a key parameter. A simple convolution model can be used to calculate an Impuls-Response-curve (IRC), based on observed precipitation surplus and river flow data. The shape of the IRC can be an important indicator. The InVEST model does not allow to perform an assessment of base flow stability and water buffering capacity within the catchment.

But predicting the impact of specific measures on river flow metrics is not evident. Firstly, this can only be done by using fully distributed physically based hydrological models at a high spatial resolution, and these are difficult to develop, calibrate and apply. Secondly, these hydrological models are designed and calibrated to predict the response of flow and groundwater levels to weather scenarios, not land management scenarios. Thirdly, the result depends very much on the assumptions that are made concerning runoff, erosion, evapotranspiration and these may not be represented in a realistic manner when working at coarse spatial resolutions. So there is need to assess and model these properties in a very detailed manner when we want to asses impact of measures.

Measures that alter hydrological pathways to improve the available water quantity and quality are all supporting this final service. Therefore, it is more useful to explicitly assess the contributing factors, rather than the final result. For this reason, PROWATER aims to spatially explicit quantify opportunities to improve water retention on field and landscape level, rather than predicting the catchment scale impact on flows. The latter may be done by using the PROWATER results to develop more accurate parameter input files for already existing hydrological models. But we need to be aware that hydrological responses at the catchment scale to land management measures will be difficult to observe in the face of uncertainties in the hydrological data and the variability and change in climate variables over the period of the records (Hess et al. 2010). In short, hydrological models are calibrated against hydrological data that encompasses a period of changes in land use, river morphology and weather patterns (climate). Therefore, it is often more useful to assess and monitor the response to particular weather events at a smaller scale. This can give insight on the relative importance of different hydrological pathways (infiltration, runoff, drainage, seepage, evaporation).

To also make the impact of water abstraction explicit, we also need to model and assess trade-offs of water abstraction on other ES. Abstraction from groundwater alters groundwater levels and associated ES. Abstraction from rivers and reservoirs affects ecological flows. This can in turn also affect other ES.

1.4. Integrated models for ecosystem service assessment

1.4.1. Requirements for ecosystem service assessment

ES assessments can facilitate the sustainable use and protection of biodiversity, ecosystems and natural resources (Carpenter et al. 2009). An ES-analysis allows to demonstrate the full spectrum of societal benefits that can be delivered by EbA measures. It can be used to optimize the spatial allocation and design of the measures. An integrated analysis of the multiple aspects of the ES-cascade for each ES is desirable and requires the integration of both biophysical and socio-economic aspects. This means that both supply and demand are incorporated in the ES-assessment (Boerema et al. 2017).

Seppelt et al. (2011) collected the four essential components of a holistic ecosystem research. These include:

1. Biophysical basis: the assessment of multiple ES requires the interactions between ES functions and therefore one needs to understand comprehensively the biophysical reality. As mentioned further on, the biophysical representation can be incorporated in models in with various degree of complexity.
2. Trade-offs: they happen when ES respond differently to changes. Turkelboom et al. (2015) defines them as “a situation where the use of one ES directly decreases the benefits supplied by another”. They can be observed on the same place and in a different area for different ES and between the present and future use of the same ES. Analysing multiple ES requires the consideration of synergies and trade-offs.

3. Off-site effects: decisions on one location can affect the delivery of ES on another location.
4. Comprehensive stakeholder involvement, which is important to link the ES functioning to human well-being.

1.4.2. Integrated ecosystem service modelling tools

There exists a large variety of methods and models to perform ES-assessments. Following Martínez-harms & Balvanera (2012) these methods can be divided in three approaches: benefit transfer approaches where a monetary value is applied to the land cover based on other site-based studies (1), community values methods combining social perceptions with biophysical data (2) and modelling ES-supply using social and ecological variables (3). Below is a brief overview of some of the most known ES assessment models, more specifically independently applicable and generalizable landscape modelling tools. They are used to quantify, map and perform economic valuation of ES in a spatial explicit manner (Sharps et al. 2017). They include biophysical processes because ES are not only determined by land-use (Staes et al. 2017), can model multiple ES and can be used for scenario analysis and decision-making. However, they can differ in approach, underlying assumptions, resolution and scale (Sharps et al. 2017). A review of other ES tools that quantify and assess ES can be found in Bagstad et al. (2013) and Neugarten et al. (2018).

InVEST stands for Integrated Valuation of Ecosystem Services and Trade-offs and is an example of ES modelling tool using socio-ecological data. It uses a production function approach, based on the supply-service-value chain, to quantify and value ES making use of models for 18 individual ES, of which 12 relate to terrestrial and freshwater systems and 6 to the marine environment. It makes use of biophysical information to estimate multiple ES in across a landscape. The output exists of ES service and monetary value maps. The purpose of InVEST is to compare scenarios, but it can also predict trends of ES provision (Nelson and Daily 2010; Neugarten et al. 2018; Sharp et al. 2018; Sharps et al. 2017). In contrast to ARIES (see below), the model parameters have been defined in advance, so that the user just have to specify the parameters. Customization of the models is possible in order to obtain the desired outputs. It also requires little specialised expertise (only GIS) which makes it easy to apply. A drawback might be that most of the models require data of the user in an appropriate format (Neugarten et al. 2018).

LUCI (Land Utilisation and Capability Indicator) also uses biophysical data (e.g. elevation, land cover, soil data) to estimate ES supply. It is also an example of ES modelling tool using socio-ecological data. It acts as a decision support tool and detect locations where land use changes might improve the supply of ES. In addition it includes a trade-off tool (Bagstad et al. 2013; Nelson and Daily 2010; Sharps et al. 2017).

Lastly ARIES (Artificial Intelligence for Ecosystem Services) is an open source modelling platform and uses a benefit transfer approach. The ES provision of each point in the landscape is based on its land use and land cover (LULC) and based on information from other site-based studies. Different models can be built and integrated by means of machine learning and context-specific data, which allows fitting the model structure to diverse application contexts. ARIES includes spatio-temporal dynamic analysis and probabilistic analysis (Bayesian networks) to obtain differences in potential and actual benefits (Bagstad et al. 2013; Nelson and Daily 2010; Neugarten et al. 2018; Sharps et al. 2017; Villa et al. 2014). Limitations of ARIES are the required specialised expertise of the users to develop, specify models and parameters and provide the necessary data. Therefore ARIES is not suitable for rapid assessments as it is time and data-intensive (Neugarten et al. 2018).

Although the tools can assess multiple ES, the tools actually comprise single-service models, lacking interactions between models of different ES. Unfortunately, these models are not able to model feedbacks and interactions that represent spatial and temporal dynamics. Furthermore, these tools do not capture spatial and temporal processes that generate the observed patterns of ES today. That way saturating or cumulative effects are not revealed (Rieb et al. 2017; Seppelt et al. 2011).

An overview of the different models and ecosystem services covered by the abovementioned ES modelling tools can be found in Table 1.

Table 1: Overview of ecosystem services modelled by different modelling platforms (Bagstad et al. 2011; LUCI 2018; Sharp et al. 2018)

InVEST	ARIES	LUCI	ECOPLAN
Carbon	Carbon sequestration and storage	Agricultural production	Agricultural production (net revenue)
Coastal Blue Carbon	Aesthetic viewsheds and proximity	Erosion risk and sediment delivery	Timber production
Coastal Vulnerability	Flood regulation	Carbon sequestration	Energy from biomass harvest (from natural sources)
Crop Pollination	Subsistence fisheries	Flood mitigation	Water provision (abstraction)
Crop Production	Coastal flood regulation	Habitat provision	Pollination
Fisheries	Sediment regulation	Water quality – Nitrogen and Phosphorus	Water infiltration
Habitat Quality	Water supply		Water retention
Habitat Risk Assessment	Recreation		Carbon storage biomass
Marine Fish Aquaculture			Carbon storage soil
Offshore Wind Energy			Nutrient storage soil (N/P)
Recreation			Nitrogen removal (denitrification)
Reservoir Hydropower Production			Erosion prevention
Scenic Quality			Air quality regulation (particulate matter)
Sediment Retention			Noise mitigation
Urban Cooling			Urban heat island mitigation
Urban Flood Risk Mitigation			Recreation (visitors)
			Impact on property value
			Health effects (DALYs)

1.5. Targeted ecosystem-based adaptation measures

There is evidently a wide suite of potential measures that enhance the residence time of water and nutrients within a landscape. With PROWATER we focus on specific types of measures that: (1) improve soil permeability through agricultural soil management, (2) reduce interception through forest conversion/management practices, (3) promote and prolong water storage in floodplain wetlands, (4) promote deferred infiltration through restoration of upstream depressional wetlands and (5) remediate soil sealing impacts through infiltration ponds.

Spatial prioritisation methods for these Ecosystem-based adaptation (EbA) measures were reviewed in the first report of WP2 of PROWATER (deliverable 2.1.1). 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. The PROWATER approach makes use of a multiscale topographic position index (TPI) to define priority areas for the application of measures. The tool developed based on this topographic indicator distinguishes landforms with distinct hydrological functions (e.g. recharge zones, permanent wetlands, temporary wetlands, runoff retention zones, ditches). Depending on the context (soil/topography/geology) and the type of interventions/measures, the user can focus on either small-scale runoff driven processes or more large-scale groundwater flow driven processes. Combining both scales is useful, as at the large-scale infiltration seepage patterns depict main recharge areas while the small-scale patterns depict local opportunities for erosion control and runoff collection. Retaining water in small scale landscape depressions can sustain downstream base flow for longer periods and promote groundwater recharge through deferred infiltration.

Depending on the combination between soil, landscape position and land management, there are several options to enhance infiltration and retention.

In the next section different key-measures that will be implemented in the context of PROWATER are explained. The final objective is to quantitatively assess the impact of these EbA inspired restoration scenarios on ES.

1.5.1. Reducing interception through forest conversion

Interception is the process where rainfall is captured in the canopy and evaporates back into the atmosphere. This water is not actively used in any biological process. Interception and forest cover in general have strong positive effects on heavy soils as it buffers extreme precipitation events, reduces runoff and promotes infiltration. The interception losses are minor compared to the runoff losses a sparsely vegetated soil would generate. On sandy, well-permeable soils, the opposite occurs. These soils are unlikely to generate runoff and interception losses reduce groundwater recharge.

During the 19th century, large areas of mixed deciduous forest in Western Europe were converted to productive coniferous forest (Verstraeten 2013). This has had serious impact on the water balance of the landscape. Changes in forest cover affects water yield, runoff, infiltration and evaporation and therefore groundwater recharge (Allen and Chapman 2001). Several studies prove in general that coniferous trees consume more water than deciduous trees. This can be attributed to higher evaporation and interception compared to deciduous hardwoods (e.g. Adane et al. 2018; Brown et al. 2005; Dams et al. 2008; Filoso et al. 2017; Nisbet 2005).

Especially pine and fir have very high interception and therefore reduce the infiltration capacity. In Flanders, such forests have often been planted on the more elevated parts within the landscape. These ridges have predominantly dry and sandy soils that were unsuitable for agricultural exploitation. These sites are also very important for groundwater recharge as groundwater levels are deep and remote from draining streams. Forest conversion to broadleaf forest or more open vegetation types (Figure 8) allows building up additional groundwater head during winter, which mitigates impacts of droughts.

Therefore, we will assess the interception in canopy and litter layer for various vegetation types. The interception ratio is also highly affected by the precipitation patterns themselves. Throughfall of additional precipitation will be 100 % once the canopy (resp. litter layer) is saturated.

We also look at transpiration losses, but in less detail. Without transpiration, there would be no primary productivity and no cooling or cloud formation. Therefore, we only look at vegetation types (and crops) that are alien for the 2 seas region and have excessive transpiration losses.

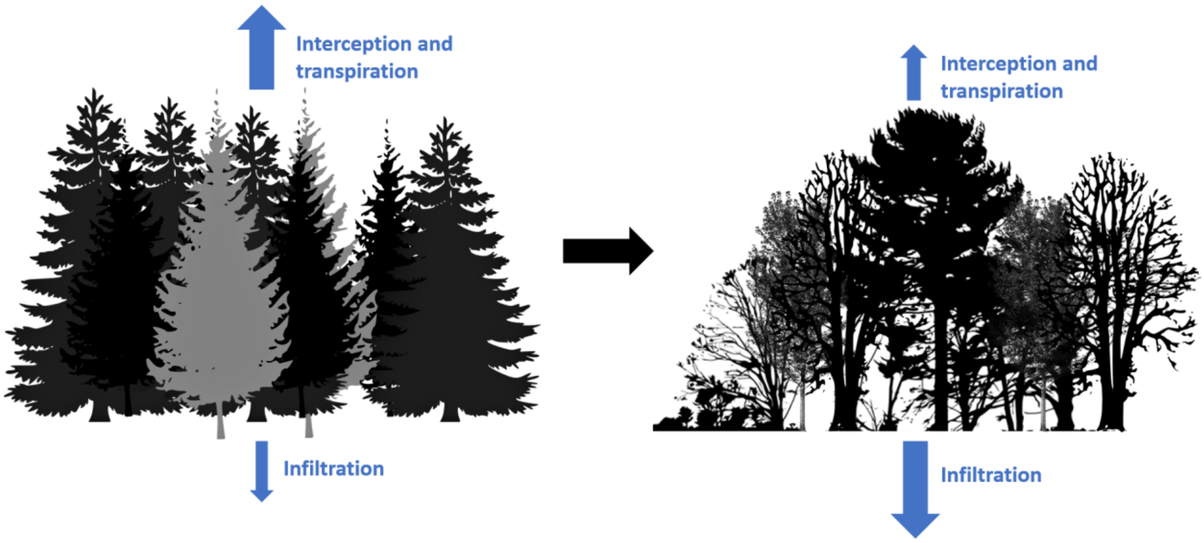

Figure 8: Illustration of conversion from coniferous forest to deciduous forest which leads to higher infiltration capacity due to lower interception and transpiration of the deciduous trees.

Heathland is an oligotrophic habitat type with a characterizing flora and fauna. People started converting these areas in more economic profitable land uses, for example arable land, at the end of the 18th century. It suffers mostly from acidification through N deposition, originating from agriculture and traffic, and also from drainage for moist heathland types. One of the techniques to restore heathland (Figure 9) is topsoil removal to restore the nutrient poor conditions and expose the seedbanks that still exist after many decades (Aerts et al. 1995; Frouz et al. 2009; Kooijman et al. 2016). This technique leads in general to a decrease in soil organic matter and nutrients (in particular N mineralization) (Aerts et al. 1995; Kooijman et al. 2016).

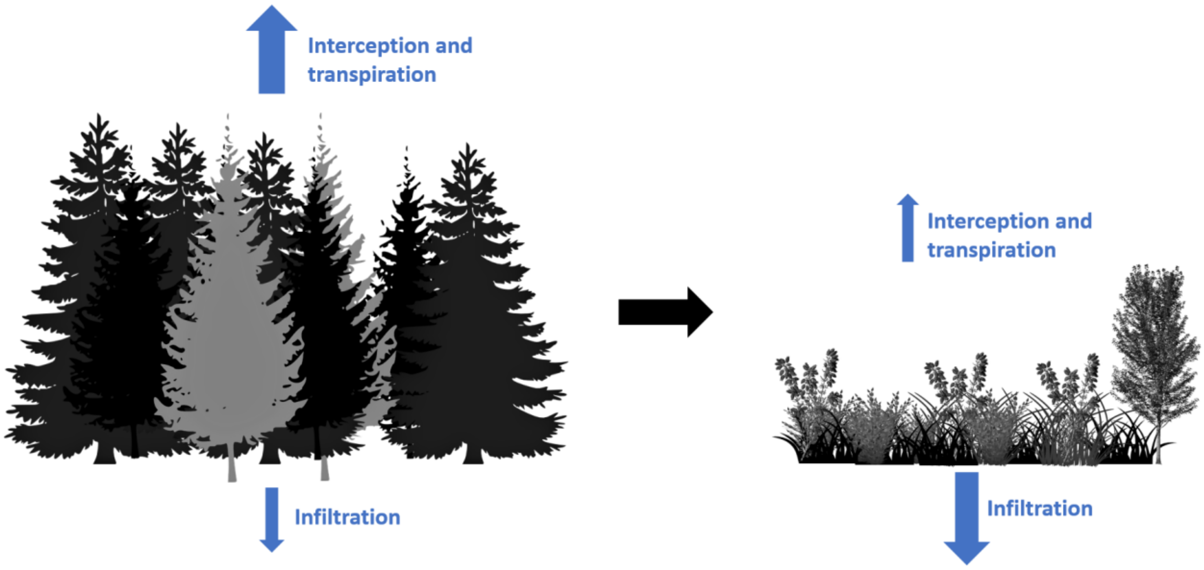

Figure 9: Conversion from (coniferous) forest to heathland leads to higher groundwater recharge due to the higher infiltration rate of heathland (lower interception and transpiration).

Another aspect to take into account concerning interception and infiltration (groundwater recharge) is the identification of the optimal tree cover density. Trees have several benefits as they store rainwater and enhance infiltration, but on the other side interception and transpiration occur. Ilstedt et al. (2016) introduced the optimum tree cover theory whereby an intermediate tree density maximizes groundwater recharge (Figure 10). Surface runoff and evaporation are high in case without trees while dense tree cover leads to high interception and transpiration. In these both cases groundwater recharge is limited. Intermediate tree cover assures maximum groundwater recharge without affecting the additional benefits of trees. This concept should be taken into account when analysing the effect of forest and heathland conversion. The optimum tree cover density can be used in the model in ECOPLAN to quantify groundwater recharge.

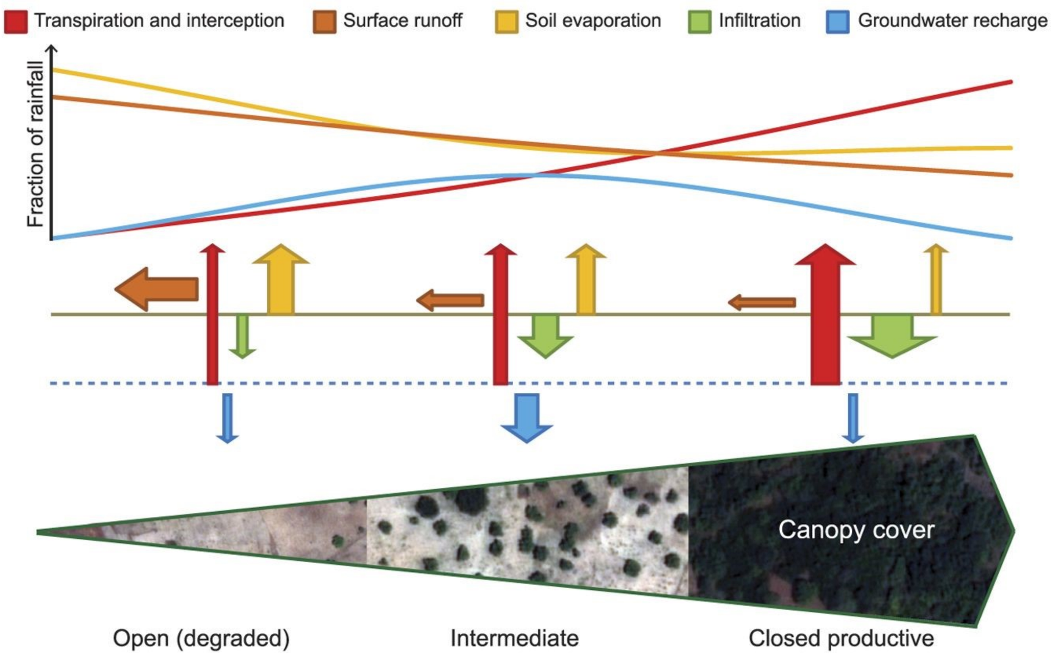

Figure 10: Illustration of the optimum tree cover theory (Ilstedt et al. 2016)

1.5.2. Improving soil permeability through soil management practices

Soils are a key asset in protecting and restoring the ability of a catchment to provide clean water. Soil can absorb, store and hold water before allowing it to drain to groundwater, rivers or be taken up by plants. The ability of soils to infiltrate and store water depends on soil texture (for example sandy soils have higher infiltration rates than clay soils) and structure (the way that soil particles are combined or aggregated). Soil organic matter content (SOM) plays an important role due to its impact on soil structure. Soil cover and water content are additional characteristics that influence soil water infiltration and retention.

In practice, a huge range of factors influence soils and their behaviour, including weather, altitude, depth, drainage, and past treatment. While general conclusions can be drawn on infiltration rates based on soil types, the specific context and situation will determine behaviour, for example reduced infiltration rates on sandy soils due to slaking or increased infiltration rates on dry clay soils due to cracking.

The soil is able to store and release water slowly over time and therefore capable of buffering variations in precipitation surplus. As infiltration is affected by soil structure, processes that lead to the deterioration of soil structure, namely a loss of larger macropores, will affect infiltration. Soil retention capacity also depends on pore size distribution. Soil degradation changes soil characteristics significantly and reduces soil water retention capacity. Compaction, loss of soil biodiversity, diffuse contamination by saline water and surface sealing also affect infiltration adversely. Reduced infiltration may cause an increase in overland flow and the risk of downstream flooding because of the reduced time lag between rainfall and peak flow. Soil water storage is limited in case of compaction, loss of soil biodiversity and loss of SOM (Defra 2015).

Soil compaction and soil erosion are considered as two most costly and serious environmental problems caused by conventional agriculture (FAO 2003). These two processes create excessive runoff and reduces infiltration. Soil compaction (Figure 11) is the increase of bulk density or decrease in porosity of soil due to externally or internally applied loads, often caused by agricultural machinery, grazing of livestock, timber harvesting and industrial activities (Batey 2009; Hamza et al. 2011; Troldborg et al. 2013). Compaction can occur at the topsoil and subsoil level. Soil texture, bulk density and water content are key factors in determining a soil's vulnerability to compaction. Subsoil damage and compaction is difficult and costly to remediate once it occurs (Jones et al. 2003). Deep soil compaction predominantly occurs when wet soils are trafficked. The higher the clay content, the higher the soil capacity to bear higher stresses at higher initial water content without reaching severe compaction state. The initial water content plays an important role in clayey and loamy soils. In contrast, for sandy soils, the mechanical parameters were less dependent of initial water content but more related to the initial bulk density (Saffih-Hdadi et al. 2009). Compaction can adversely affect nearly all physical, chemical and biological properties and functions of soil. Soil compaction causes a decrease in large pores (called macropores), resulting in a reduced permeability to water and air, much lower water infiltration rate into soil, increased surface runoff, erosion, reduced groundwater recharge, as well as a decrease in saturated hydraulic conductivity. Vegetation suffers from restricted rooting depth, reduction in nutrient uptake and the formation of waterlogged or anoxic zones, which can lead to denitrification and slow nitrification (Batey 2009; Cui et al. 2010).

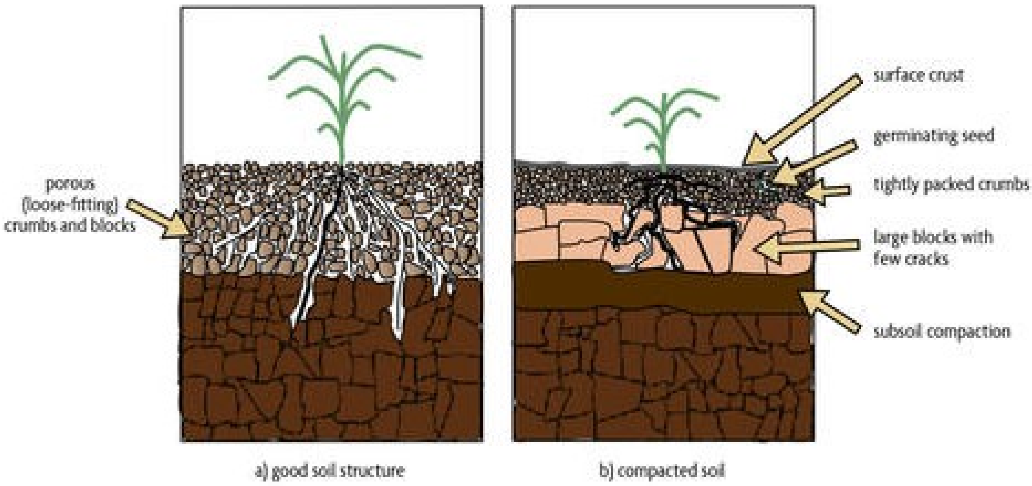

Figure 11: Illustration of soil compaction (SARE 2012)

But also shallow soil compaction affects soil functions, especially when soils have a low soil organic matter (SOM) content and are poor in forming soil aggregates that ensure a certain soil porosity (Galdos et al. 2019). SOM plays an important role in nutrient retention and provision to plants as well as soil particle aggregation, and so influences aeration, structure, drainage and other functions. SOM can hold up to 20 times its weight in water and makes soils more drought (Dick and Gregorich 2004; FAO 2005). Soil organic carbon (SOC) content has an influence on the soil structure and thus on soil compaction. SOC is a (measurable) component of soil organic matter. It enables aggregation of soil particles and ensures a stable soil structure. It prevents soil compaction and erosion and the high porosity allow a better permeability, which is beneficial for water retention and infiltration (FAO 2017). Therefore, soils with a higher SOC content are preferred. Degradation of SOM influences infiltration and retention ability of soil, and is driven by oxidation (linked to tillage practices and increasing temperatures), reduced accumulation or physical removal (removal of residues from fields and erosion) (FAO 2005).

Ecosystem-based Adaptation measures, such as non-reversing tillage, compost amendments, catch crops and crop residues, can be implemented to restore and protect soil health with an aim to support infiltration and retention of water and practices relating to soil health are already included in agri-environment schemes. These EbA practices will improve the infiltration and water holding capacity of the topsoil, but is unlikely to improve deep infiltration when the subsoil is already compacted (tillage pan). Therefore, prevention or reversal of subsoil compaction is prerequisite. The use of controlled traffic farming only compacts a small proportion (< 20%) of the field. Trafficking should be avoided on very wet and dry soils, but also the use of less heavy machinery and adapted wheel surface and pressure can help to reduce deep compaction. When compaction already occurred, deep ripping techniques (up to 1 m depth) may restore the soil (Sinnett et al. 2006). At the landscape level, impact of (subsoil) compaction can be reduced by implementing EbA measures such as retention ponds (see paragraph 1.5.5 and Figure 13), leaky dams, hedges and compartmented ditches which can act as retention and infiltration hotspots.

Agricultural management is decisive for soil health. Conservational agriculture (CA) is preferably to be chosen over conventional tillage. CA includes reduced or no tillage and minimal soil cover by retaining crop residues (Busari et al. 2015; Palm et al. 2014). No till management (in combination with residue retention) leads to an increased bulk density in the topsoil layer and reduces soil porosity. Increased organic matter and biotic activity leads to an increased soil aggregates stability and greater macropore connectivity due to a higher abundance of macrofauna. These processes lead to an increased infiltration and reduced runoff and erosion (Busari et al. 2015; Congreves et al. 2015; D'Haene et al. 2008; Galdos et al. 2019; Palm et al. 2014; Tebrügge and Düring 1999; Verhulst et al. 2010). The impact of CA on soil chemical properties remains less clear because this is less studied, but studies point to higher SOC, improved N use efficiency, lower N losses and a higher nutrient availability in the top soil (Palm et al. 2014; Verhulst et al. 2010). In addition, it is shown that reduced tillage may reduce the global warming potential of CH_4 and N_2O by 6.6% (Feng et al. 2018). Table 2 summarises the effect of no tillage on soil physical, chemical and biological properties.

Table 2: Impact of no-till (NT) compared to conventional tillage (CT) (first column) and surface residue retention vs no surface residue placement (second column) on soil biological properties and processes (a), soil physical properties, processes and ES(b) and soil chemical properties, processes and ES (c) (Palm et al. 2014)*. This information originates from Verhulst et al. (2010) and was summarised by Palm et al. (2014).*

a.

Soil biological properties and processes	NT compared to CT	Residue retention
Soil organic matter in topsoil	↑	↑
Particulate or labile organic matter fractions	↑	↑
Soil microbial biomass	↑	↑
Microbial functional diversity	↑	↑
Fungal populations	↑	↑
Enzymatic activity	↑	↑
Beneficial micro-organisms (fluorescent Pseudomomas; Actinomycetes, some Fusarium strains)	↑	↑
Pathogenic micro-faunal: Take-all Gaeumannomyces; Rhizoctonia, Pythium, and Fusarium root rots	↑	↑
Free-living (beneficial) nematodes	ns	↑
Plant-parasitic nematodes	↓	ns
Earthworms	↑	↑
Arthropod diversity	↑ more so for predators then phytophagous arthropods	↑

b.

Soil physical properties, processes and ecosystem services	NT compared to CT	Residue retention
Aggregate stability	↑	↑
Bulk density	↑ but small number of studies showing opposite	↓
Total porosity	↓	↑
Macropores	↓↑ avg size larger	↑
Mesopores	↑	
Micropores	↑	
Hydraulic conductivity	↓ mixed results	↑
Infiltration	↑	↑
Runoff	↓	↓
Evaporation	↓	↓
Plant available water	↑	↑
Erosion	↓	↓

c.

Soil chemical properties, processes and ecosystem services	NT compared to CT	Residue retention
Total nitrogen	↑ follows pattern of soil organic matter	↑
Nitrogen availability (N mineralization)	generally ↓ at least in the short term and often long term	↑↓ depends on quality of residues
P, K, Ca, Mg	P ↑ in top soil layer. K ↑ in surface layers, in general. Ca, Mg few differences	K depends on type of crop residue
Cation exchange capacity	no effect	↑ but only in very top layer
pH	more often ↓	↓
Nutrient leaching	??	??

Furthermore catch/cover crops can be used to improve infiltration and soil health in general. Cover crops grow during the fallow periods. They have the potential to protect the soil from erosion, reduce nitrate leaching and losses of nutrient, pesticides and sediment, increase soil organic matter and carbon sequestration and reduce pest and weed pressure. Leaf cover prevents physical degradation of soil aggregates and decayed plants roots form channels (De Baets et al. 2011). They therefore improve water infiltration and water and soil quality (Basche et al. 2014; Dabney et al. 2001; Kaspar and Singer 2011). A summary of the advantages is shown in Table 3 (Dabney et al. 2001).

Table 3: Advantages and disadvantages of the utilisation of cover crops (Dabney et al. 2001)

Advantages	Disadvantages
Reduce soil erosion	Must be planted when time (labor) is limited
Increase residue cover	Additional costs (planting and killing)
Increase water infiltration into soil	Reduce soil moisture
Increase soil organic carbon	May increase pest populations
Improve soil physical properties	May increase risks of diseases
Improve field trafficability	Difficult to incorporate with tillage
Recycle nutrients	Allelopathy
Legumes fix nitrogen	
Weed control	
Increase populations of beneficial insects	
Reduce some diseases	
Increase mycorrhizal infection of crops	
Potential forage harvest	
Improve landscape aesthetics	

Although most literature attributes generally positive effects of conservation agriculture (reduced to zero tillage), the use of catch/cover crops and increased soil organic matter, there are a considerable number of studies that show no effects or even negative effects on plant growth. It is clear that one cannot transform from one management practice to another and expect immediate effects. It takes time for the soil ecosystem to mature to a new equilibrium. A lot depends on the state of the soil before measures are taken. We can assume that applying no-till on heavily degraded soils with low SOM will have negative effects. Firstly, because subsoil compaction is likely to be present and secondly because the topsoil is very vulnerable to physical degradation. Therefore, it is crucial to apply deep ripping and sufficient SOM amendments (or fallow period with deep rooting vegetation) before transforming to no-till management.

1.5.3. Creation and Restoration of permanent wetlands

While infiltration and deferred infiltration in upstream areas results in a strategic long-term aquifer recharge that spans multiple seasons, there is also need for water retention in downstream valley wetlands. Under natural conditions, these valley bottom wetlands have permanently wet conditions. But most often, these valley systems are (partly) in agricultural use and heavily drained. Valley bottom wetlands act as sponges and can provide base flow during drought periods. Important measures are to decrease the drainage basis of both the drainage network and the main drain. In the past, a lot of streams have been straightened to improve drainage and parcel layout for agriculture. This river normalisation has reduced flow friction and accordingly aggravated flood frequency and magnitude. River re-meandering and floodplain restoration does not only alleviate downstream floods, but also has the potential to store a lot of water in the peaty subsoil (Figure 12). This is even more important when groundwater abstractions are present. Many abstractions from rivers and groundwater take place in the more downstream valleys.

Decreasing the drainage basis often requires re-meandering, providing more in-stream water storage and creating a more gradual riverbed slope along the floodplain. The name meander is derived from Maiandros, which in turn is derived from the Latin meander, meaning wandering or wandering around. The morphology of a river largely depends on the (spatial) distribution of flow velocity in the bed. Even within the same bend in the meander a typical flow pattern arises in which the water in the outer bend will flow faster than at the inner bend. These differential flow velocities in meandering rivers cause an asymmetric bed in the meander curves. Meanders can be subdivided on the basis of their sinuosity. The stronger the river meanders, the higher this factor. A perfectly straight river has a winding factor of 1, a normal meandering river has values between 1.5 and 2 and an extreme meandering river has values above 3.

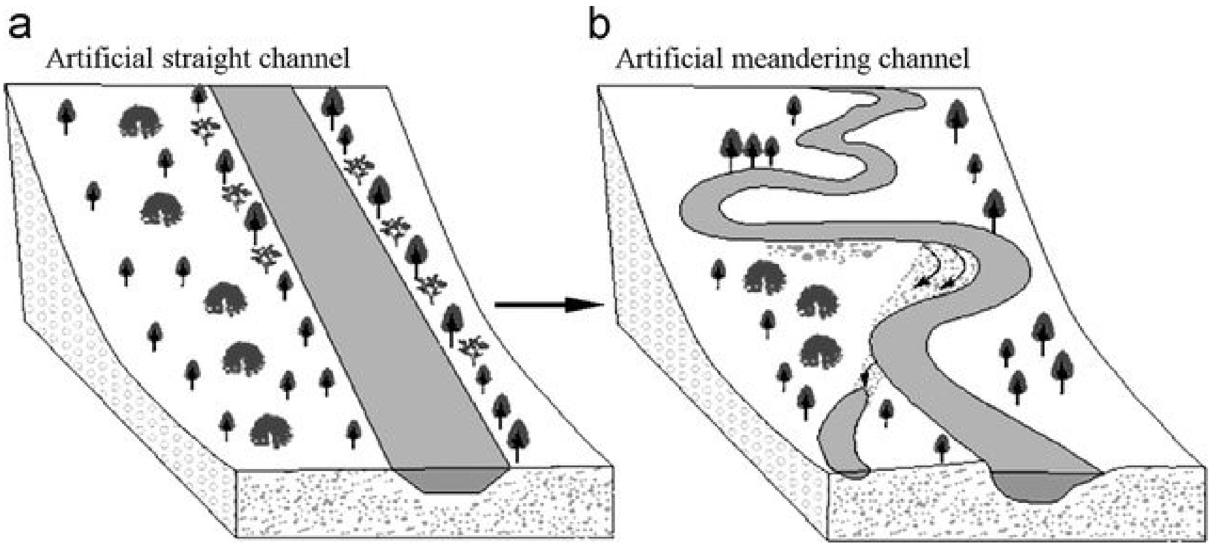

Figure 12: Diagrammatic sketch of restoration of the river meandering characteristics (Pan et al. 2016)

Re-alignment of channelled pathways results in a significant increase in sinuosity. In addition, the groundwater level may rise as a result of the increased route length, allowing weirs to be removed. The smaller attenuation of the new remodelled course results in a higher flow velocity, despite the less steep slope than canalised watercourses. The increase in river length will cause a general decrease in river slope and therefore an increased flood frequency and inundation of the alluvial plain. Usually the flow rate will decrease, especially in the most upstream areas of the restored watercourse.

The most drastic consequences of a change in a watercourse, whether natural or forced, are an increase in its total length, in other words a reduction in the suspension S of the watercourse and an increase in total friction. As a result, peak flows at a certain geographical location in the river will be more flattened, which will reduce the risk of flooding at that location.

1.5.4. Creation and Restoration of temporary wetlands

Wetlands play an important role in the hydrological cycle (Bullock and Acreman 2003) and provide numerous environmental functions (Bertassello et al. 2018; Mitsch and Gosselink 2000). There is growing evidence that small scale wetlands play a disproportionally large role in regulating hydrology (Bertassello et al. 2018; Colvin et al. 2019). A number of recent papers specifically focus on the flow regulating functions of wetlands that are not hydrologically connected to the river network. Different terminology is used in the literature to refer to such wetlands, namely “depressional wetlands” (Evenson et al. 2018), “non-floodplain wetlands” (Jones et al. 2019; Lane et al. 2018) or “geographically isolated wetlands” (Cohen et al. 2016; Evenson et al. 2016; Rains et al. 2016). Due to their topographical position, these areas are naturally characterized by a high fluctuation in water levels (hydroperiod with short lag time, high frequency, low amplitude). This creates possibilities for deferred infiltration which recharges groundwater reserves and increases base flow during subsequent periods of drought (Lee et al. 2018). Available studies show that the actual groundwater recharge by wetlands depends on the interplay of buffer volume, retention time and hydraulic conductivity of the subsoil.

Despite recognizing the importance of hydrological function of wetlands, basin-scale wetlands services have rarely been investigated (Wu et al. 2020). The representation of (small) wetlands in catchment models is a known issue (Evenson et al. 2016; Sharifi et al. 2016). This caveat is only recently being addressed. Results from a study in North-East China revealed that when wetlands are properly represented exert significant impact on basin hydrological processes by decreasing streamflow and altering streamflow regime (magnitude, frequency, duration and time of flow events).

Although their importance is now recognised, small scale temporary wetlands have long been viewed as problematic in terms of agricultural production and, consequently, have been subject to land drainage or infilling (Acreman and McCartney 2009). 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 runoff dynamics. Periods of excess precipitation can lead to temporarily waterlogged conditions. Centuries ago, most of these wetlands were drained by drainage channel networks (Staes et al. 2009), 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.

1.5.5. Runoff collection, storage and infiltration

The concept of deferred infiltration in natural landscape depressions can be expanded to the creation of artificial infiltration ponds that collect runoff from compacted soils and impervious areas in heavily modified landscapes. They differ from temporary wetlands, which we consider as natural elements in the landscape due to their topographic position. Runoff collection, storage and infiltration facilities are constructed elements in the landscape and are generally much smaller in size than temporary wetlands. But when designed as EbA measure, they can evidently deliver a lot of services.

Both imperious surfaces following urbanisation and degraded agricultural soils cause alteration of hydrological processes, such as increased surface runoff, soil erosion and subsequent floods in downstream areas (Gill et al. 2007; Guerra et al. 2019; Korgaonkar et al. 2018). Restoration of the hydrological functioning can be achieved by reversing soil sealing and decompacting soils. Since this is not always feasible and cost-effective, we can achieve the same effect through active management of (storm) drainage systems and actively creating retention-infiltration ponds (Figure 13). In regions with high groundwater exploitation, a large scale deployment of distributed storm water collection facilities that collect and infiltrate excess hillslope runoff before it reaches a stream, could make a big difference in groundwater recharge (Beganskas and Fisher 2017). Once the capacity of the facility is reached, increased runoff may occur, especially if these facilities are sensitive to clogging (Gette-Bouvarot et al. 2014). Therefore, it is important to design facilities that have sufficient capacity or emergency storage, especially considering climatic change and extreme precipitation events.

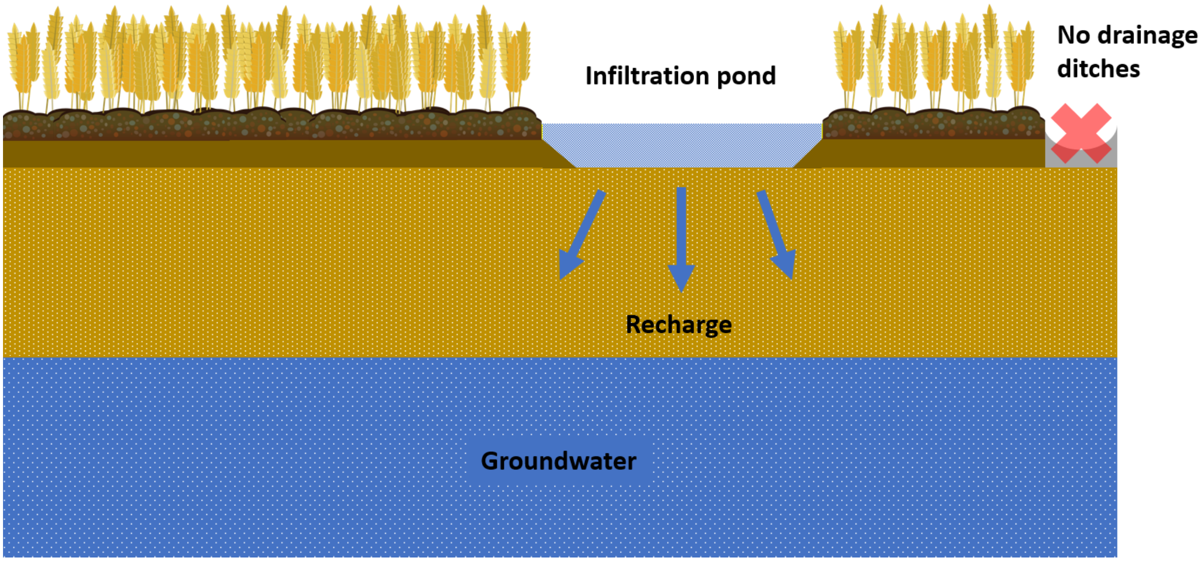

Figure 13: Illustration of an agricultural infiltration pond. Drainage ditches are not recommended as they prevent groundwater recharge.

The use of infiltration techniques has therefore increased and are frequently used in an urban and rural context for enhancing water retention and infiltration and influencing various ES, such as water provisioning through water storage and groundwater recharge, water purification, flood mitigation and erosion control. In addition, they can be visually attractive and reduce societal costs. However, their main disadvantages include clogging of suspended load and contamination of the topsoil layer and of the groundwater (Bardin et al. 2002; Dechesne et al. 2004a, 2004b; Gette-Bouvarot et al. 2014).

Superficial infiltration of runoff from rooftops is feasible in almost all situations. Especially for soils overgrown with grass and herbs, the top layer is sufficiently permeable and will dry out quickly. In principle, no complicated infiltration device is required for the drainage of a gutter. A few metres of drainage away from the facade and allowing it to drain out is sufficient. A very shallow depression on the lawn is sufficient. The infiltration zones will expand when extreme precipitation occurs. Such small-scale systems for individual roofs from private individuals are possible in almost all circumstances if sufficient space is available (Figure 14).

Figure 14: Rooftop runoff infiltration system (left) and a collective water retention and infiltration facility (right).

The construction of lowered verges and infiltration depressions is recommended for runoff from road surfaces or larger roof areas of multiple homes. These are very effective for infiltration, especially if the surface area is sufficient, it is overgrown and there is little treading. The use of pebbles or wood chips will increase the infiltration capacity of the top layer. Especially in urban areas, this also offers opportunities to create urban green space. There is little chance of saturating the subsurface with infiltration water. There is little chance of silting up and damaging the infiltration capacity.

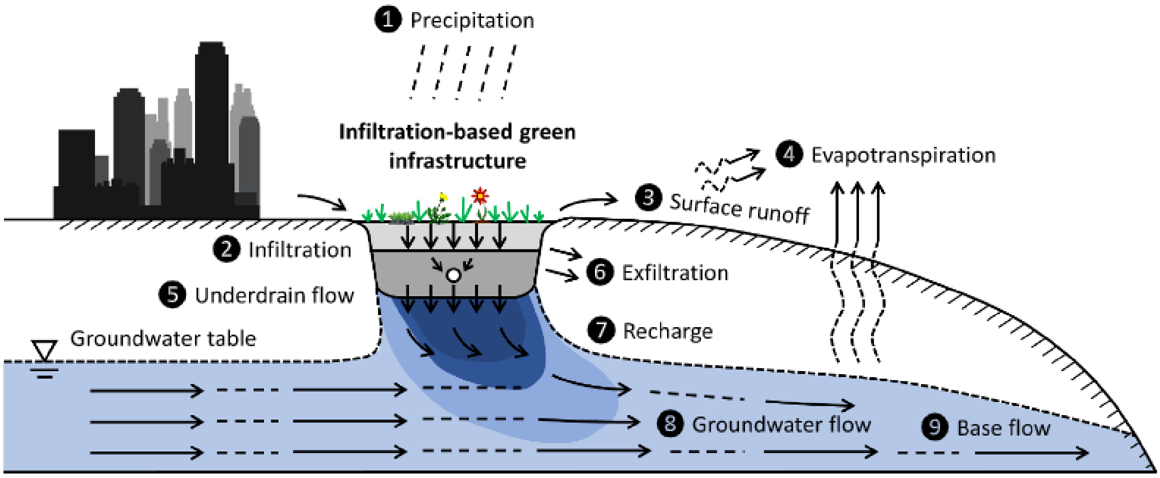

Figure 15: Schematic representation of the main hydrological processes involved in the interaction between infiltration-based green infrastructures and shallow groundwater (Zhang and Chui 2019)

In case of open shallow ditches for storm water drainage along roads, simple weirs (Figure 16) can be installed to maintain higher levels and increase groundwater recharge in upstream areas (Maliva 2020). In addition they increase nutrient and sediment retention (Dollinger et al. 2015; Kröger et al. 2008; Kröger et al. 2011). Scientific studies show that it is important that the soil of such drying canals is overgrown. Wide shallow overgrown canals are more effective than deep narrow canals. At the narrow deep canals, sediment and sediment will clog up the bottom of the ditch. On the one hand, vegetation at narrow deep canals has limitations due to the light climate at the bottom of the ditch. On the other hand, silting up will cause further stagnation and sedimentation of particles, which will cause vegetation to die off. Clearances may be necessary to remove the silted soil, but this should preferably be done at the end of the winter so that vegetation has a chance to colonise the soil. Studies show that even high levels of sediment deposits in the lower areas do not pose a problem for the infiltration capacity, as long as it can remain overgrown. The vegetation provides a better distribution of sedimentation and the grass roots create macropores in the soil (Ahmed et al. 2015).

Figure 16: Creating compartments in sloping ditches for storm water drainage enable slow infiltration and groundwater recharge. Without these weirs, the water would be drained too quickly to infiltrate.

Despite of their great potential and widespread use, infiltration techniques such as infiltration basins and permeable surfaces are often not explicitly included in hydrological models due to the mismatch in spatial resolution. Most catchment scale hydrological models are lumped models and even distributed models often have pixel sizes that are considerably larger. While measures can be implicitly included by altering the imperviousness at pixel level, it does not allow to assess the contribution and benefit of particular measures. With PROWATER, we will focus on quantifying the impact of specific measures on runoff/retention/infiltration at the landscape level.

2. Review of ecosystem services quantification methods

In this section for every abovementioned ES we will provide an overview of ES quantification methods with respect to the context of PROWATER. The most important principles, methods and their feasibility, accuracy and data requirements will be discussed. A selection of these methods is used in the ECOPLAN-SE tool, developed by the ecosystem management research group (ECOBE) of the University of Antwerp. The ECOPLAN-SE (scenario evaluator) is an existing spatial explicit GIS-tool that has been developed through the ECOPLAN project. It is used to calculate the provision of different ES based on spatial scenarios in terms of changes in land use and landcover. The ES calculations are carried out using models resulting in quantitative ES maps (Vrebos et al. 2017). Lastly, the impact of infiltration and retention measures on ES and how these measures can be integrated in ECOPLAN-SE will be assessed.

2.1. Water regulation

Although water regulation is often labelled as a particular ecosystem service, it is actually an umbrella term for many different ecosystem services related to hydrology of catchments and ecosystems. Therefore, it is more appropriate to label water regulation as an ecosystem function. Ecosystem services are those aspects of ecosystem functions to which we attribute a particular societal value. Services are often benefit dependent (Boyd and Banzhaf 2007), meaning the benefits you are interested in will dictate what you understand as an ecosystem service. For example, water regulation services are an intermediate input to the final service of clean water provision. Water regulation is particularly sensitive to interpretation and consequently, there are various interpretations to be found in literature. Water regulation for which purpose and at which scale are pertinent questions.

So which final benefits are associated with water regulation? For a farmer, water regulation will be relevant at the field scale an relate to minimising waterlogged conditions or a good water holding capacity of the soil to overcome droughts. But water regulation can also refer to preventing floods and droughts. To recharge aquifers? To improve base flow? To avoid erosion? But these very different objectives are not always achieved simultaneously by measures and trade-offs exist.

Even the recharge of aquifers and their particular role in providing a strategic water security is a very different ES than merely the annual abstracted volume and its market value. This particular focus on strategic water security is important for PROWATER, but is seldom recognised and difficult to express in monetary terms.

So water regulation has many facets. We will first review the more general integrated assessments of water regulation as an encompassing concept (Table 4). These approaches are often applied on a catchment scale. Most often they relate to the catchment water budget. Thereafter we describe the various components of water regulation and their interactions.

Literature review

Climate change will affect the hydrological cycle and the spatial and temporal distribution of precipitation. In particular the seasonality of water availability and extreme hydrologic events will have a substantial effect. Chang and Bonnette (2016) summarized the impact of climate change on hydrological ES in a cascade: climate change has an impact on temperature (minimum, maximum, variability, range) and precipitation (timing, amount, intensity, form) which in turn affect the hydrological system in terms of changes in evapotranspiration, soil moisture and timing and amount of runoff. As a result, water yield may be affected.

Table 4: Overview of available tools to quantify the ES of water regulation.

Tool	Parameters	Reference
	One indicator	
Water storage potential	Annual aggregated soil filtration (capacity of terrestrial ecosystems to store surface water), annual subsurface flow (service flow of water regulation-	Liquete et al. (2011)
Water infiltration	Infiltration rate	E.g. Benegas et al. (2014), Le Clec'h et al. (2016), O'Farrell et al. (2009), Smukler et al. (2010), Van Eekeren et al. (2010)
	Composed indicators	
Water retention potential	LAI, total area of surface water bodies, total available water content, lithology, slope, degree of soil sealing	Vandecasteele et al. (2018)
Hydric significance indicator	Precipitation amount, slope, soil types according to the wetness index, soil texture, ground transmissivity, land use and forest ecological status	Šatalová and Kenderessy (2017)
	Modelling tools	
SWAT	Biophysical modelling of water regulation	Arnold et al. (1998)
WATYIELD	Biophysical modelling of water regulation	Fahey et al. (2010)
INVEST: water provision module	Computation of water yield	Bagstad et al. (2011)
ARIES: water supply module	Computation of hydrological variables	Sharp et al. (2018)

There are studies that just make use of one indicator for water regulation for rapid assessment. For instance Liquete et al. (2011) included both the capacity to provide as well as the actual flow of services which are quantified in biophysical terms on European scale. An estimation of the capacity allows to assess potential changes in the service provisioning. The water storage potential is used as an indicator for water regulation. The capacity of terrestrial ecosystems to store surface water was estimated by the annual aggregated soil filtration and for their service flow of water regulation is the annual subsurface flow used as the indicator. These simple indicators provide a general view on the provision of water regulation, albeit on a larger scale.

The ecosystem function of water infiltration has been frequently used to assess the water regulation and provision capacity by means of direct field measurements (e.g. Benegas et al. 2014; O'Farrell et al. 2009; Smukler et al. 2010; Van Eekeren et al. 2010). Water regulation has also been quantified by the infiltration rate as by Le Clec'h et al. (2016). They assessed the infiltration rate by field measurements of 135 sampling points which is then linked by statistical regression to land cover data in order to map this particular ES-indicator. They mention that this indicator can be used on different spatial scales.

Several studies develop approaches to quantify water regulation with an indicator comprising of several biophysical factors. Vandecasteele et al. (2018) for instance developed the Water Retention

Index (WRI) to assess the water regulation potential of the landscape at European scale. This water retention capacity is subdivided into different components as listed below. These six components are combined in one indicator to estimate the water retention capacity on a larger scale.

- Vegetation retention is dominated by interception and is positively correlated with the Leaf Area Index (LAI). For the WRI, LAI was estimated from land use maps.
- Retention in surface water bodies: the total area of surface water bodies is used as a proxy to estimate this component in the WRI.
- Soil water retention: the capacity of the soil to retain water is a function of soil texture, bulk density and organic matter. The finer the soil texture, the higher the water holding capacity will be, except for clay rich soils. Also, the amount of organic matter is positively correlated with the soil water retention potential. The WRI makes use of the total available water content derived from the European Soil Database to estimate the basic level of soil water retention and changes in organic carbon content.
- The main explanatory factor for groundwater retention used in WRI is lithology. Its permeability will determine the infiltration rate. Values for the WRI are derived from literature.
- Also, the slope is an important factor, as the retention capacity will be higher in case of gentler slopes.
- Finally soil sealed surfaces allow little or no water retention. The degree of soil sealing for the WRI is derived from land use maps.

Such indicators also exist on a smaller scale. Šatalová and Kenderessy (2017) estimated the water retention function of the landscape by means of the hydric significance indicator. This indicator comprises of seven factors: precipitation amount, slope, soil types according to the wetness index, soil texture, ground transmissivity (the ability of the aquifer to transmit water), land use and forest ecological status. A weight was given to each of the factors according to their impact on the water retention capacity of the landscape. Land use is assumed to be the dominant variable for water retention, however also precipitation is determining but is not taken into account. In comparison with the previous study this indicator can be assessed on a regional scale. Individual properties can be set on a high resolution (e.g. 10 m).

The ES of water regulation and provision has also been quantified by making use of modelling. The Soil and Water Assessment Tool (SWAT; Arnold et al. 1998) has been used to model all water cycle processes (ecosystem functions) in the watershed and generate detailed outputs in space and time that can be used for ES analysis, including water provision, regulation and purification and erosion regulation. It models for instance nutrient and pesticides fluxes through the watershed. The input comprises of all kinds of environmental parameters (soil, climate, land use, topography, land management). The water basin is divided into subwatersheds and further into hydrologic response units (HRUs) which are combinations of unique land-use, soil and slope. However, this detailed input data is not always available and learning the model costs time. Also interactions with organisms are not captured (Logsdon and Chaubey 2013; Vigerstol and Aukema 2011).

Another tool is the WATYIELD water-balance model (Fahey et al. 2010) used for modelling the water balance in small water basins (from 1 km² to 5 km²). It models the daily water fluxes of rainfall, interception, evapotranspiration and drainage related to a soil profile. Input data include daily rainfall and potential evapotranspiration and the output comprises of the daily runoff. If you combine this for different land covers, soil types and climates, the model generates the mean annual water yield. Disadvantages are the data intensity and the time-consuming process of data gathering and model running (Dymond et al. 2012).

SWAT and WATYIELD are examples of discipline-specific models and are only developed for a specific domain, in this case the hydrologic cycle or balance of a watershed (Rieb et al. 2017). Additionally, all the above-mentioned indicators and models focus only on the biophysical supply of water provision and regulation. Other models like InVEST and ARIES (see also 1.4.2) also account for the benefits for people. InVEST is used to model several ES to illustrate general pattern and changes in ES as a result of land use changes or climate change impacts. The model of water provision calculates the water yield (precipitation minus evapotranspiration) for each cell and is based on the Budyko Dryness index (Budyko 1974) and annual average precipitation. Required input data include topography, land use, soil data, plant available water content and plant evapotranspiration coefficient (Bangash et al. 2013; Terrado et al. 2014). Key drawbacks are the rather lower resolution (200 m) and seasonal variability, groundwater and water resource infrastructure are not taken into account (Vigerstol and Aukema 2011). Also only one ES can be simulated at the time, so comparing interactions between ES is difficult (Logsdon and Chaubey 2013). Furthermore, the hydrological processes are rather simplified and this represents uncertainty (Vigerstol and Aukema 2011). Lastly, InVEST is often used in studies to quantify ES provision in single catchments or sub-catchments within a single river basin. In contrast to these studies, Redhead et al. (2016) executed a general validation of the InVEST model for water provision across 22 catchments in the UK. They compared modelled water yield with input parameters from national scale and global scale datasets and general validation was done by empirical measurements of gauged daily water flow. This study shows that InVEST modelling results in a good performance, but local input data and parameters are preferred so they represent the spatial and temporal scale. InVEST does not account for points abstractions, such as for water reservoirs (important for water supply in the UK) which encompass large volumes of water, but can be represented as LULC classes with conform consumptive water use values in case they can be obtained from the water industry.

Water supply can be modelled with ARIES where the available input data is used to compute hydrological variables, such as precipitation, infiltration and evapotranspiration, taken into account soil and vegetation data). In contrast with InVEST, the benefits to the users are also quantified (Martin-Ortega et al. 2015) and trade-offs between provision in ES can be assessed. The models can also operate on small cell sizes (e.g. 30 m), depending on the input data. One of the drawbacks include the lack of transparency of the model code because of its complexity (Vigerstol and Aukema 2011).

Current representation within ECOPLAN-SE

There is currently no catchment scale integrated assessment of water regulation available within the ECOPLAN-SE. Water yield is determined by a cascade of water losses (interception, runoff, infiltration, evapotranspiration and so on. Many of these hydrological pathways are already available within the ECOPLAN-SE, but not all of them and they have not been summarised/integrated at the catchment scale.

Ambitions for PROWATER

In the next sections, we will review the different aspects of water regulation, briefly explain the quantification methods that are currently available within the ECOPLAN-SE and propose targets for improved representation within PROWATER.

As already depicted in figure 3 and 4, we aim to develop detailed spatial explicit quantification for each aspect of the potential hydrological pathways. In Figure 17, the different pathways are further detailed.

We will develop methods to map and assess each of these hydrological pathways. The PROWATER measures are explicitly targeted to impact these specific pathways and the geophysical potential for implementation is depicted by the spatial prioritisation tool. The spatial prioritisation tool depicts key groundwater recharge zones for which we need to reduce interception and improve topsoil infiltration capacity, suitable zones for runoff collection and infiltration at field scale, locations of (former) temporary wetlands and at the largest scale it identifies (former) permanent wetlands. All measures target to increase the residence time of water within the catchment and promote deep groundwater recharge. Manipulating evapotranspiration by vegetation is not targeted as an EbA strategy to increase water availability. Evapotranspiration by vegetation is a natural pathway that we don't consider problematic. In case of invasive alien species or deliberate plantations with excess impact on water resources, such measures are legitimate (e.g. Eucalypt plantations). But then measures should be focussed on specific species. But this is not the case within the 2 SEAS region. In order to be complete, the evapotranspiration box was added to Figure 17.

Finally, we will also propose an overall assessment method at the catchment scale. It is not our ambition to precisely predict water volumes for each of these pathways and predict exact water budgets. We do not ambition the development of a fully distributed hourly time step hydrological model as we focus on the impact of Nature-based Solutions. Especially because there are a large number of unknown factors. We do not incorporate or assess the impact of technical measures (e.g. water transfers across catchment).

We do aim to evaluate the difference that land, soil and water management can make and how this can increase aquifer storage, increase base flow, decrease peak flows and improve the performance of many water related ES such as water quality, carbon sequestration, crop yield and so on. The high-resolution assessment allows to make the impacts of land, soil and water management measures tangible to land planners and land managers. This is an important difference from hydrological models, which often neglect the impact of land management in their modelling approaches and are often unable to distinguish the impact of land management changes from natural variability and unknown factors. Ultimately, PROWATER's high resolution layers could be summarised to lower resolution input layers for hydrological models.

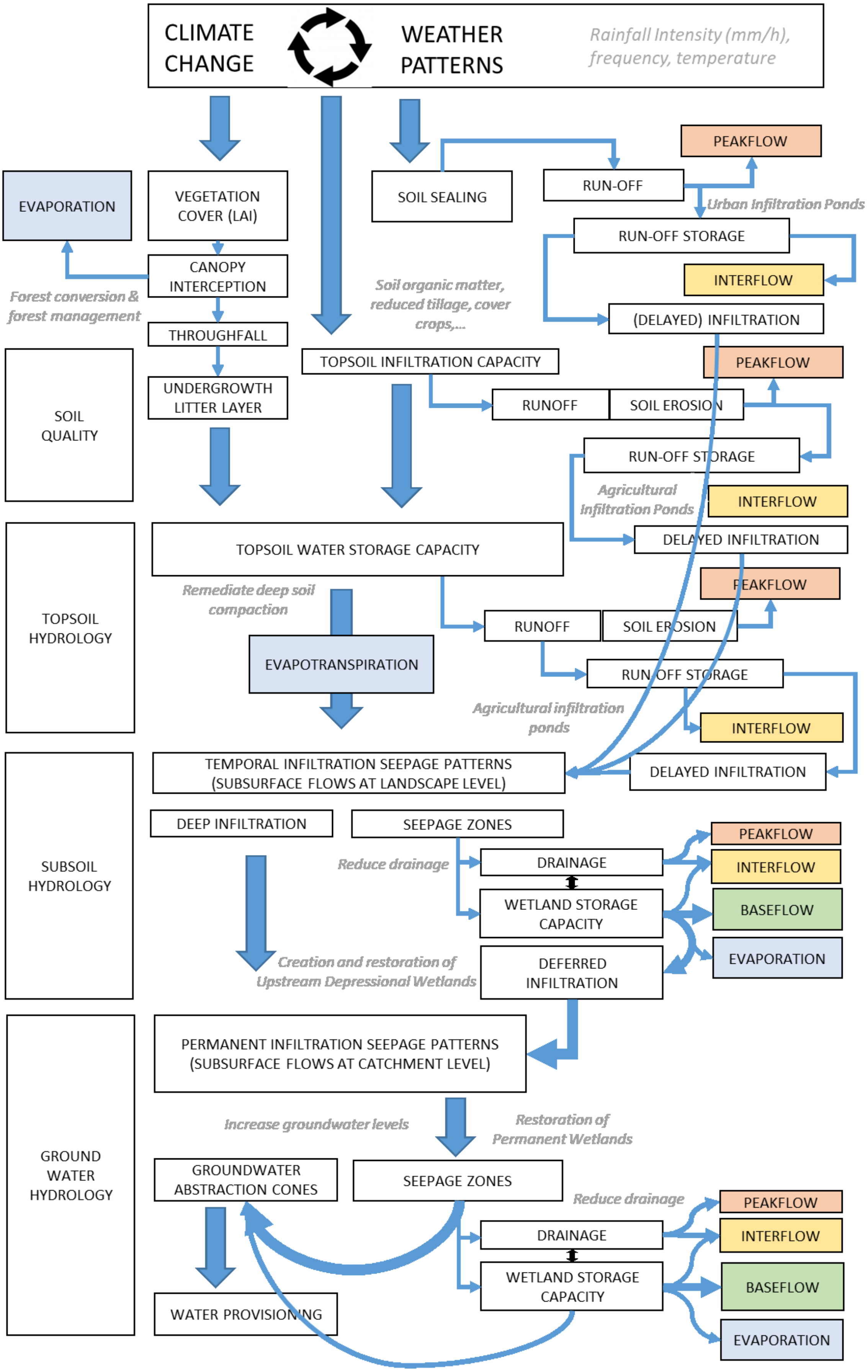

Figure 17: Schematic representation of the hydrological pathways that are addressed by PROWATER

2.1.1. Infiltration and interception by vegetation

Literature review

The capacity of the landscape to effectively store and infiltrate precipitation is dominated by vegetation interception and evaporation. These two processes explain the difference in groundwater recharge between forests and less dense vegetation types (e.g. grasslands, heather). Only the proportion of precipitation that reaches the soil layer can be stored, drained or infiltrated. This potential can be changed through land use conversion and soil management. Forest cover plays an important role in water retention (EEA 2015). In particular conversion from coniferous forest to heathland has proven to be beneficial as already mentioned in the introduction. But there are many nuances to be made.

Interception loss is greatly dependent on the canopy cover, which determines the amount of infiltration that reaches the soil and this depends on vegetation type (Figure 18 and Figure 19). There is a major difference in interception between coniferous and deciduous forests, but there are also more subtle differences between vegetation types. Major factors are leaf life span, leaf area index, leaf morphology and tree crown morphology (Yang et al. 2019).

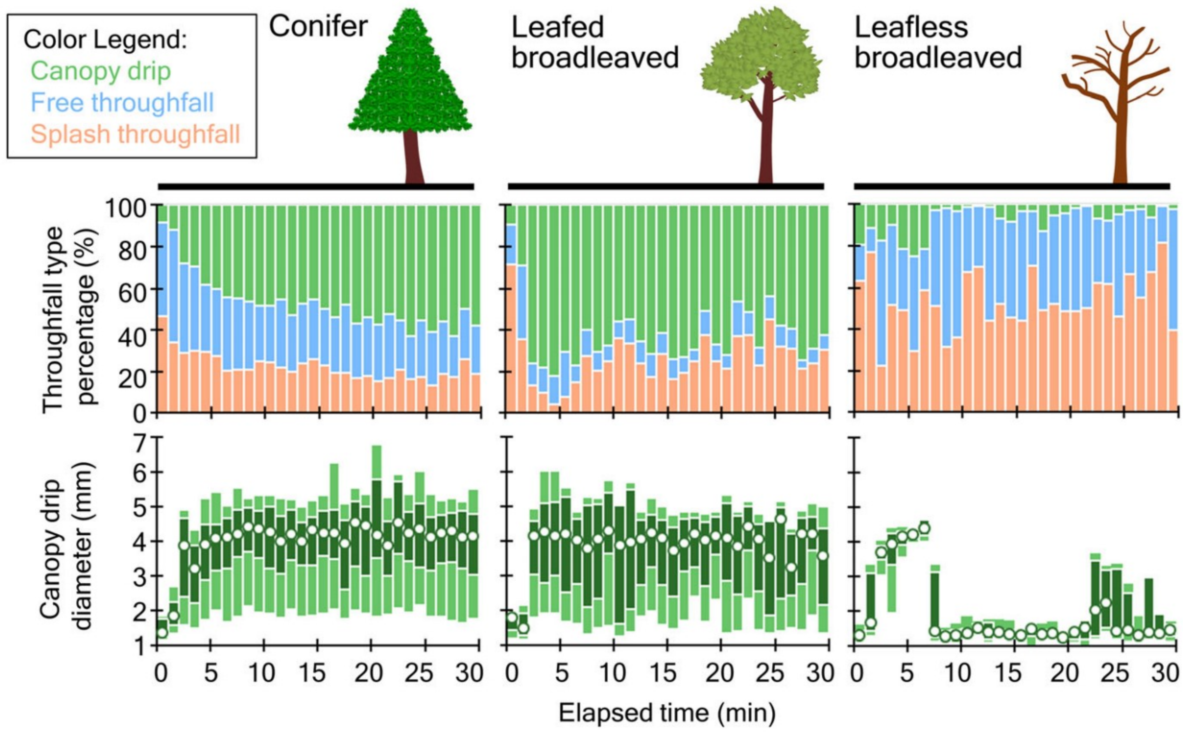

Figure 18: Temporal variation in the volume percentage of throughfall types and drop diameter of canopy drip among three tree groupings under a simulated rain event with an intensity of 20 mm hr−1 : coniferous species, leafed broadleaved trees, and leafless broadleaved trees. Drop diameter is shown by box and whisker plots with respective cumulative drop volume percentiles Dxx (10%, 25%, 50%, 75%, and 90%) (Levia et al. 2019).

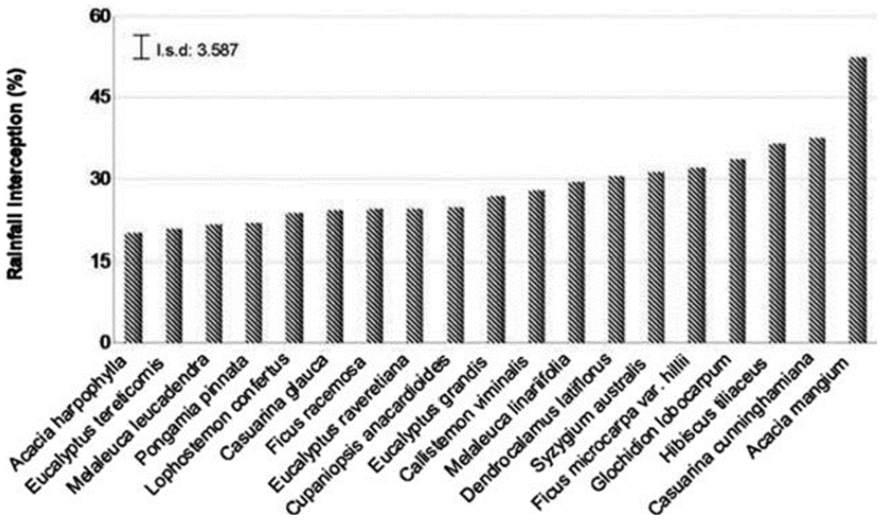

Figure 19: Mean interception for different vegetation species (Hester 1996)

In general it is shown that evaporation and interception are higher in forests than in the case of shorter vegetation. Calder et al. (2002) found in their study on the potential impacts on water resources of proposed afforestation that in general groundwater recharge follows the following rank order: grassland > heath > oak woodland > pine forest (Table 5). Several studies have also proven in general that coniferous trees consume more water than deciduous trees because of higher evaporation and interception (Brown et al. 2005; Dams et al. 2008; EEA 2015; Filoso et al. 2017; Nisbet 2005). Figure 20 shows the difference in interception ratio for deciduous and coniferous hardwoods in the UK (Nisbet 2005), while Table 6 represents estimations of interception for several forest types in Hungary (Gribovszki et al. 2019).

The impact on conversion from one land use type to another is dependent on the soil type, scale of planting, forest design and replaced landcover. The above-mentioned rank order is applicable for sandy soils, but the situation is different for chalk soils. Conversion from grass to broadleaved woodland on chalk soils has a little impact as the uptake of root water can be maintained, even during drought periods (Calder et al. 2002; Nisbet 2005). The water use of grass is also dependent on the management (Nisbet 2005).

The difference between grass and broadleaved forest might be limited for lighter-foliaged trees, such as ash, however a reduction in groundwater recharge is still expected to be the case on drought-prone soils and in wetter areas (The Woodland Trust 2008). Interception of heather can be higher than that for deciduous forests, but this is partly counterbalanced by its lower transpiration rate.

The forest design includes age and the size of the woodland. The higher evaporation at the edge of the forest results in higher interception losses, but this effect is limited to less than 20 m from the edge and is therefore only crucial for woodlands that are small in size (< 1 ha). Furthermore the species composition also influences this edge effect (Calder et al. 2002; Herbst et al. 2007; Klaassen et al. 1996; Nisbet 2005).

Table 5: Typical ranges of annual losses for transpiration, interception and total evaporation (mm) for different land covers in case they receive 1000 mm of annual rainfall (based on several studies in UK) (Nisbet 2005).

Land cover	Transpiration	Interception	Total evaporation
Conifers	300–350	250–450	550–800
Broadleaves	300–390	100–250	400–640
Grass	400–600	–	400–600
Heather	200–420	160–190	360–610
Bracken	400–600	200	600–800
Arable*	370–430	–	370–430

*assuming no irrigation.

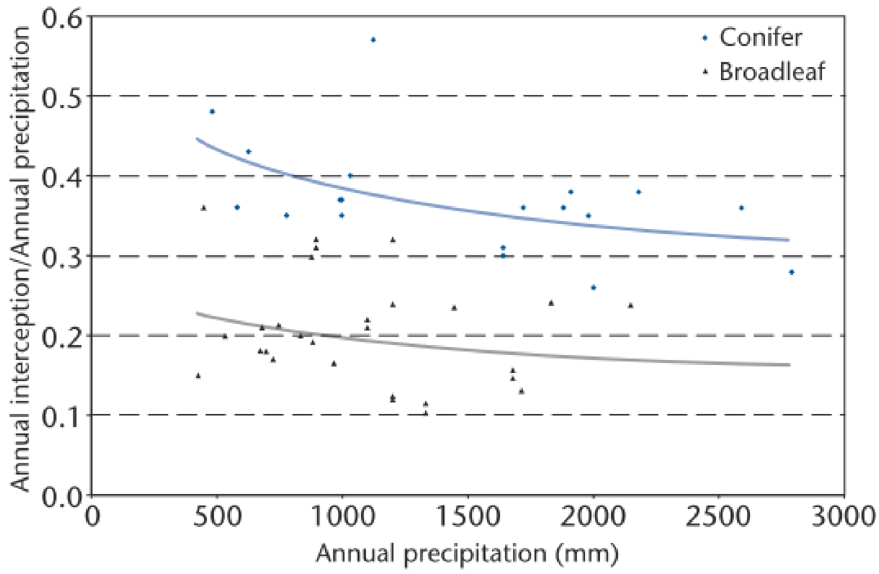

Figure 20: Studies in the UK show that between 25% and 45% of the total annual rainfall is intercepted by coniferous hardwoods, compared with 10-25% for deciduous hardwoods (Calder et al., 2003; Nisbet, 2005).

Table 6: Results of interception studies on Hungarian forests (interception expressed as a percentage of the annual precipitation) (Gribovszki et al. 2019)

Forest type	Interception [%]	Source
Turkey oak – Sessile oak	22.3	*Szabó* (1979)
Sessile oak	25	*Führer* (1984)
Hybrid poplar clones	25	*Járó* (1980)
Hornbeam –Turkey oak	27	
Turkey oak	27.5	
Linden	28	
Hybrid poplar clones	29	
Black locust (young)	30	
Beech (young)	30.9	*Kucsara* (1998)
Black locust (old)	31	*Járó* (1980)
Northern red oak	33	
Beech (unmanaged)	39.7	*Koloszár* (1981)
Larch	34	*Járó* (1980)
Scots pine (young)	35	
Balck pine (young)	36	
Scots pine (old)	37	
Spruce (old)	37	*Führer* (1984)
Eastern white pine	36	*Járó* (1980)
Hornbeam-Scots pine	37	
Douglas fir	38	
Black pine (old)	39	
Spruce (young)	41.6	*Kucsara* (1998)
Spruce (middle age)	40.5	*Kucsara* (1998)
Beech (managed)	47 28 29.7	*Járó* (1980) *Führer* (1984) *Koloszár* (1981)

Rainfall interception in temperate forests typically ranges between 9 and 48% of gross precipitation, but is highly dependent on the total rainfall, including the duration and frequency of rainfall events (Sheng and Cai 2019). Intensity/duration/frequency (IDF)-relationships describe the relation between rainfall intensity, the duration of this rainfall event and the frequency with which the combination of these two previous parameters occurs (Du et al. 2019). These relationships are commonly used in water resources engineering (e.g. for the design of stormwater facilities) (Koutsoyiannis et al. 1998; Willems 2000). An example of IDF-relationships is shown in Figure 21.

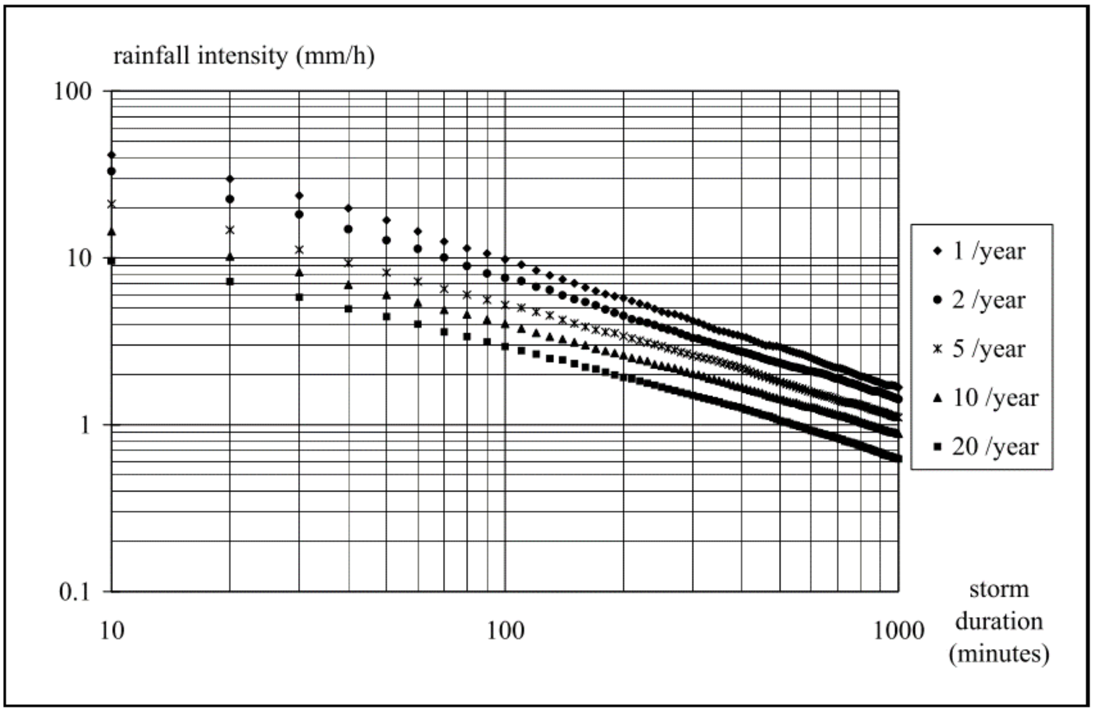

Figure 21: IDF-relationships for Uccle (Belgium) (Staes et al. 2011; Willems 2000, 2011).

This explains why relative and absolute interception can vary significantly for the same vegetation type and is generally lower in regions or years with higher precipitation (Figure 20). Therefore, it is better to look at metrics that allow to calculate the effect of precipitation intensity and variability. "Surface water storage capacity" (SWSC) or "Canopy storage" (CS) of vegetation types indicate the interception potential of a single rainfall event. Evidently, a high SWSC is associated with high interception, but it is a more objective measurement and can be used to assess the potential impact of changes in IDF as a consequence of climate change. Moreover, we can also quantify its positive effects on heavy soils, where the canopy storage buffers and prevents soil runoff and erosion.

Leaf area index (LAI) and vegetation area index (VAI) are two commonly used metrics that can be related to canopy storage (CS) and thus interception. SWSC is widely used in models and is determined by the surface morphogenesis of tree and other vegetation species. Coniferous species have in general higher SWSC values than broadleaf species. The capacity is also dependent on rainfall intensities and duration and can therefore be assessed by means of IDF-relationships. SWSC increases with rainfall intensities and plateaus after a certain intensity. Xiao and McPherson (2016) found out that interception decreased with increasing duration and frequency of rainfall events. They concluded that interception controlled differently according to the duration of a rainfall event: (the distribution of) precipitation plays a dominant role for smaller rainfall events when SWSC is determining for larger rainfall durations.

The method commonly used to derive this canopy storage capacity is based on the work of Leyton et al. (1967). The most commonly used methods for calculating CS of a forest canopy are the model from Rutter (1967) and the analytical adaptation of this model by Gash and Morton (1978). Their conceptual models include gross precipitation, crown storage, throughfall, stemflow and evaporation (Smets et al. 2019).

The study by Smets et al. (2019) shows that for northern maple and small leaved lime, interception storage for small rainfall events (< 5 mm), canopy storage is about 40 % (3 mm), while during heavy rainfall events (20 mm) only 23-28% is captured (5 mm). For events between 5-10 mm, small leaved lime has larger interception storage (57 % - 5.7 mm) than northern maple (40 % - 4 mm). In another study on spruce in China (He et al. 2014) throughfall equals $0.72 * P_{event} - 1.16$ (with P = precipitation). This equals 2.5 mm (50 %) canopy storage for a 5 mm event, 4 mm (40 %) for a 10 mm event and 7.7 mm (38%) for a 20 mm event. Douglas fir is among the trees with the highest leaf area index (up to LAI = 11 m^2/m^3) and the highest water storage capacity (2.5 mm). But also, other vegetation types and crops can have high interception values. Mature corn (maize) can intercept 0.5 mm before throughfall occurs (Stoltenberg and Wilson 1950).

Figure 22 illustrates the relation between CS and LAI for various study sites in the Netherlands (Moors 2012). CS is expressed as the volume (mm) of water that is stored in the canopy before through fall occurs and remains available for evaporation after the rainfall event.

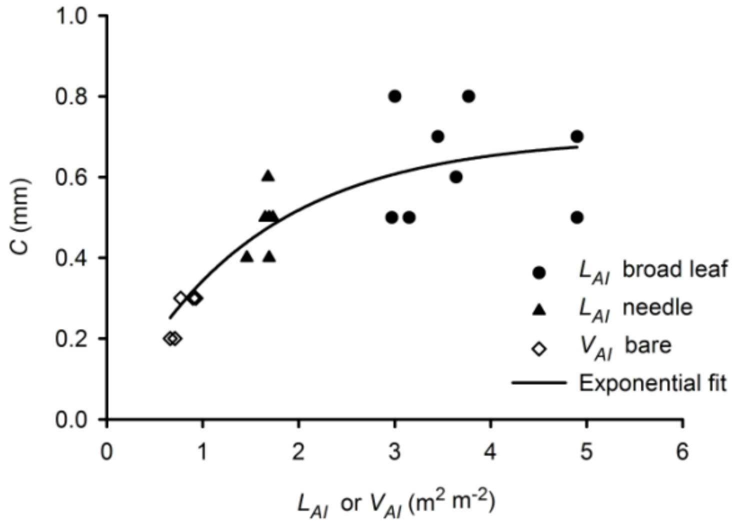

Figure 22:The water storage capacity C as derived by Leyton's analysis using daily data from the Netherlands (Moors 2012)

The regression equation $C(V_{AI}) = 0.7\ (1 - exp^{-0.67V_{AI}})\ with\ R^2 = 0.75$ describes the relation between C and the Vegetation Area Index V_{AI} (Moors 2012). For values of LAI > 3 the increase in C with increasing leaf area becomes less important. This exponential rise of C to a maximum is partly due to the fact that for LAI > 3 the canopy cover of most forests in The Netherlands is almost 100%.

Besides the leaf area, other characteristics of the tree such as the angle of leaves and branches and the distribution of the leaves also play a role in the relationship between C and LAI. For example, adding the leaf angle or the gap fraction will increase the variance explained by approximately 10% (R^2 = 0.87).

To make it even more complex, recent studies show that canopy interception is also temperature dependent (Klamerus-Iwan and Błońska 2018). Water droplets that adsorb to leaves are significantly larger for warmer rain temperatures due to different contact angles of droplets to leave tissue. This relation equates a rise following the equation in Figure 24. This can have important implications for forest hydrology and thus groundwater recharge.

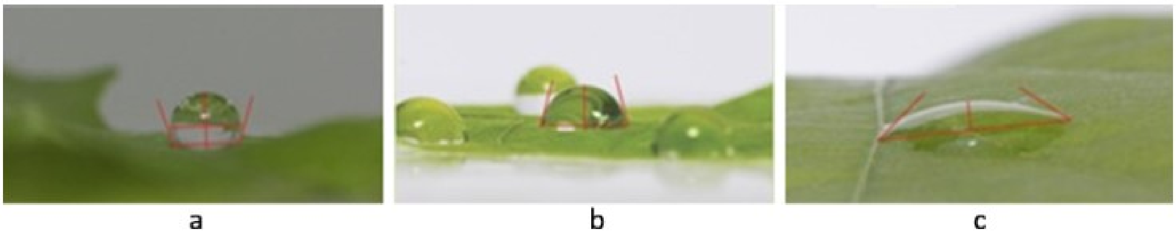

Figure 23: Images of water droplet on Oak from the shadow-free chamber at 25°C (a), at 13°C (b) and at 4°C (c) (Klamerus-Iwan and Błońska 2018)

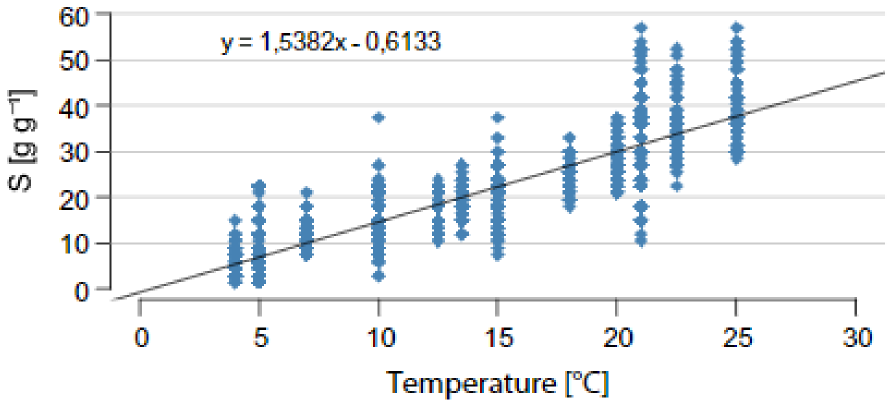

Figure 24: Effect of temperature on canopy storage capacity (S) (pooled data from all species) (Klamerus-Iwan and Błońska 2018)

The results of site-based studies have limited representability for regional scale since forest canopy is often shown to be inhomogeneous with large spatial and temporal variability in its stand distribution, LAI and canopy fractional coverage (Cui and Jia 2014). Some detailed studies on individual trees express the canopy storage in gram per unit of dry biomass or per leaf area. In absence of biomass or leaf area data, these figures are difficult to use. Studies that report interception figures rain, temperature and rainfall intensity seem to play an important role.

Overall, it is difficult to find key figures of CS or LAI per tree species because there are many factors that interplay, such as forest growth and management, tree density, stand age, presence of undergrowth.

Finally, there is a need for further research on (forest floor litter) interception. Often it is assumed that throughfall, which is the rainfall that is not intercepted by a canopy, is fully available for infiltration. This is not correct. The forest floor also intercepts rainfall much in the same way as the canopy (Gerrits and Savenije 2011). Many empirical studies on water balance in forests have disregarded this component, mainly due to the difficulty of directly measuring forest litter interception. A rare study on this topic revealed a rather linear relationship between dry litter mass and litter storage capacity (Rosalem et al. 2018) (Table 7):

$$C_{max} = 0.6771x + 0.851 \ (R^2 = 0.90)$$

$$C_{min} = 0.5384x + 0.5638 \ (R^2 = 0.96)$$

Table 7: Forest litter interception in an area of the Cerrado (Rosalem et al. 2018)

Date	Dry litter mass (kg.m^{-2})			Storage capacity (mm)	
	Min*	Mean	Max**	Cmax	Cmin
Mar/15	0.98	1.26a	1.54	1.70	1.24
Apr/15	0.72	0.90a	1.21	1.46	1.05
May/15	1.42	1.72a	2.12	2.02	1.49
Jun/15	0.60	1.16a	1.67	1.64	1.19
Jul/15	1.06	1.31a	1.69	1.74	1.27
Aug/15	2.18	2.45a	2.97	2.51	1.88
Sept/15	0.89	1.47a	2.12	1.84	1.35
Oct/15	1.38	1.73a	2.20	2.02	1.50
Nov/15	1.02	1.39a	2.29	1.86	1.36
Dec/15	1.21	1.26a	1.31	1.72	1.26
Jan/16	0.88	1.26a	1.89	1.71	1.24

But also leaf traits seem to affect interception. Also, for the litter interception, coniferous forests have the highest litter interception values. For broadleaf it seems that leaves are layered in such a way that the forest floor is providing a short path for rainwater to leave the leaf litter, hence promoting litter flow and leading to less rainwater intercepted by the broad-leaf litter than the needle-leaf litter (Zhao et al. 2019). For needle litter, the structure of the litter is more homogeneous, leading top more litter water storage. An older study demonstrated that forest floor interception storage capacity of a pine forest was 2.8 mm, which is 60 % higher compared to eucalypt (1.7 mm) (Putuhena and Cordery 1996).

Conifers are traditionally considered to be less-suitable nutrient cycling improvers and/or maintainers compared to broadleaves (Kacálek et al. 2018). In general, the litter layer in coniferous forests is more dense than in deciduous forests due to low mineralisation rates. Atmospheric nitrogen deposits aggravate litter accumulation due to acidification (Oulehle et al. 2018). Field data from Central Europe shows a steady accumulation of humus in Central European Scots pine stands, with totals up to 45 t/ha in 120-year-old stands (Hille and den Ouden 2005). While there are some short-term benefits for carbon sequestration in litter, there can be negative effects on groundwater recharge due to increased litter interception. Reduced nutrient cycling also affects tree growth on the long term.

Current representation within ECOPLAN-SE

Within the ECOPLAN-SE we use a relatively simple approach with annual interception estimates for different tree species and vegetation types (Table 8). This assumes an annual average rainfall of 800 mm of which interception is subtracted. The remaining water budget is then further redistributed between transpiration, infiltration and runoff.

Table 8: Key figures for total annual interception used in ECOPLAN-SE, assuming a total average annual rainfall of 800 mm.

Vegetation type	Intercepted	Remains
Grassland (managed)	75	725
Grassland (unmanaged)	100	700
Shrubland	150	650
Phragmites	250	550
Mixed coniferous (scots pine)	300	500
Mixed leaf forest	250	550
Poplar	200	600
Orchards	150	650
Willow	175	625
Spruce/Fir	400	400
Ash	200	600
Oak	250	550
Heath	100	700
Black Pine	350	450
Birch	200	600
Vegetables	50	750
Maize	100	700
Alder	175	625
Wheat	75	725

Ambitions for PROWATER

As already mentioned in the introduction, it is proven that coniferous trees are characterized by a higher interception compared to deciduous hardwoods. Conversion into broadleaved woods or heathland would therefore be beneficial for water provisioning. Interception is determined by the intensity and duration of the rain (Xiao and McPherson 2016). Interception occurs in the beginning of the rain event and decreases exponentially with the magnitude of the rain event (Viessman and Lewis 2003). In general one can state that interception is negligible for high intensity rainfall events (Sheng and Cai 2019; Xiao and McPherson 2016).

Incorporating rainfall intensity and frequency

The IDF-relationships could be incorporated in ECOPLAN together with the effect of interception on these relationships by means of statistics (López-Lambraño et al. 2013). This allows to quantify the effect of interception for rainfall events with a different intensity, duration and frequency and this for different vegetation types.

The IDF-relationships can also be used to estimate infiltration and runoff to evaluate the effect on tillage, soil organic carbon and infiltration basins. The study of IDF-relationships with rain events with a certain duration, frequency and intensity can be used for instance to estimate the dimensions of (agricultural) infiltration basins. This is also related to the extent of soil compaction, agricultural management (tillage system) and soil characteristics as they determine the potential infiltration capacity.

To allow the assessment of weather scenarios, we probably need to develop a classification of archetype weather situations. Typical weather patterns which can be compiled by end-users into a weather scenario. We prefer this approach over a more technical approach where we actually simulate individual rainfall events. We can also use these to create random or pre-set a sequence of events that occurred in the past, based on analysis of actual weather data. A balance needs to be found between detail and applicability, but there are a number of very recognizable daily weather patterns.

> *Examples are: Dry hot day / damp hot day / damp hot day with mild thunderstorm event 0.5-1 mm/h during 30 minutes / Cold winter day with persistent mild rain / April showers day with intermittent sun and rain (5 mm) / damp hot day with extreme thunderstorm event 3-5 mm/h during 15 minutes, and so on.*

The archetype weather patterns include information on total daily rainfall and its temporal distribution, temperature, wind, …). This allows to introduce a rainfall intensity and temperature factor (Klamerus-Iwan and Błońska 2018), which are generic for all vegetation types. In general interception is higher for high temperatures. This allows to calculate the interception loss for various rainfall events. A distinction needs to be made for rainfall events below and above the CSS. This allows to calculate the interception that was lost and the net precipitation that reaches the soil.

Incorporating surface water storage capacity

The surface water storage capacity (SWSC) is an important variable to calculate interception losses on a daily basis using archetype weather situations. The surface saturation storage capacity is defined as "the thin film of water that must wet tree surfaces before flow begins, is the most important variable influencing rainfall interception processes" (Xiao and McPherson 2016). The SWSC is preferably derived from leaf area index and vegetation area index data.

For a series of vegetation types, low and high estimates for Leaf Area Index of the canopy, undergrowth and litter factor are collected from literature and completed by expert judgment (these numbers can also be adjusted when local data is available). These basic figures should reflect the situation of dense and unmanaged vegetation types. LAI/VAI will allow to calculate the SWSC and total daily interception losses for a particular weather type.

Through corrections of LAI, we can implement the effect of forest conversion and forest management scenarios, we can then correct LAI estimates.

> *E.g. unmanaged, dense Pinus Sylvestris has a LAI of 3 m^2/m^2 and a CS of 2 mm, but forest thinning can reduce this by 30 %.*

2.1.2. Infiltration and soil sealing (runoff)

Literature review

There is no hard science needed to demonstrate that soil sealing will have an impact on soil water infiltration. Soil sealing for urban and infrastructure development constitutes the most intense form of land degradation and affects all ecosystem services (Tobias et al. 2018). Urbanisation is a major reason why soil is overlaid with impervious surfaces. Soil sealing with concrete or asphalt leads to full sealing, while semi-pervious surfaces such as concrete paving slabs, still allow partial penetration of water and air. Soil sealing has a significant impact on the functioning of soil, causing an irreversible loss of its biological functions. Soil sealing increases the risk of flooding and water deficiency. Researchers and policy makers have become aware of this fact and call for limiting development and compensating for new soil sealing with unsealing measures.

In landscape indicator studies urban land uses are generally aggregated to one class, but impervious areas are often heterogeneous and generally consist of roads, houses (or roofs), industry, gardens and so on, showing it is far from a uniform land use class. Depending on the research question, for example impact of urban area on water flow, nutrients, metals, and so on, a subdivision of the class "impervious area" is required to assure a good representation of the classes.

One cannot just assume that all impervious areas are really 100 % impervious. In addition, it matters what happens to the runoff from sealed surfaces. Many roads are fully sealed, but in practice roadside swales will infiltrate a larger proportion than unpaved surfaces would.

However, if the sealed surface runoff is collected to storm drain infrastructure, there is an immediate transport to the river. Depending on this combination of imperviousness and connectivity to streams, the sealing will have higher impacts. What adds to this complexity is that storm drain infrastructure can significantly alter the catchment boundaries. Especially for combined sewer systems that drain both wastewater and rainwater. Sewer systems not only transport wastewater within the catchment, but also import and export water depending on the location of the wastewater plants. As a result, catchments are connected with (urban) satellite catchments, which are located outside its natural boundaries. In these satellite catchments wastewater, but also rainwater from impervious areas, is collected and transported to waste water treatment plants located outside the natural catchment (Vrebos et al. 2014).

The recent obligations to foresee rainwater tanks and infiltration systems (for rainwater tank overflow) adds complexity. Depending on tank size, water use and connected runoff area, there will be a certain amount of infiltration, but this is impossible to model. This setup evidently reduces the dependency on piped water but may also reduce infiltration. Only during periods with extreme precipitation surplus, rainwater tanks will overflow to the infiltration system. During extremely dry periods, most tanks will be empty. During such dry episodes, water demand is high, and people will rely on piped water. This implies that the drinking water infrastructure still needs to be built and maintained for a relatively high flow capacity. Despite a reduced consumption of piped water for low quality applications (flushing toilets, watering garden, etc.), the infrastructure cost will remain equal and will consequently result in increased water pricing. More importantly, the pressure on ecosystems due to abstraction cones will remain high during drought periods. Because groundwater reservoirs are the largest rainwater tank that exists, full infiltration of runoff water should be prioritised in key recharge areas.

Current representation within ECOPLAN-SE

The current version of the ECOPLAN-SE makes use of high-resolution land cover maps. Consequently, we can assume that paved surfaces are effectively 100% impervious, especially for rooftops.

For the Flemish Region, we have basic data on the presence of combined sewers and storm drain infrastructure. At the pixel level, we assume zero infiltration when there is sewage or storm drain infrastructure present and a potential infiltration rate when there is no such infrastructure. This potential infiltration is assumed for grassland vegetation on a particular soil type. This is an arbitrary assumption but averages out the several uncertainties. When infiltration facilities are deliberately installed, the net infiltration of paved surfaces is probably higher than the potential natural infiltration

(50-75% of total precipitation will be net recharge versus 30-40% net recharge for grassland vegetation). On the other hand, a certain proportion of the runoff will be collected in rainwater tanks and re-used. Evidently, this volume is unavailable for infiltration and groundwater recharge. Finally, not all paved runoff will be effectively connected to sewer infrastructure, even when (combined) sewer infrastructure is present.

In principle, the ECOPLAN-SE allows to specify scenarios where certain zones can be changed from pervious to semi-pervious. The assumptions on imperviousness need to be based on site specific data such as distribution and capacity of rainwater tanks and presence of infiltration facilities.

We currently only allow to restore infiltration up to the level of the potential natural infiltration under grassland. However, infiltration systems can be more effective in groundwater recharge than unsealing the soil.

Ambitions for PROWATER

For the new version of the ECOPLAN-SE, we foresee the implementation of infiltration infrastructure as a landcover type. We will model local runoff of paved surfaces and propose a certain infiltration capacity guideline. Because there are no data available, the user still needs to define assumptions on rainwater tanks and infiltration systems. But the end-user will be guided to make assumptions on setting a lower and upper limit for imperviousness.

This could be implemented in the following manner:

1) Option to select sealed surface and make them semi-permeable;
2) Designate zones as infiltration systems including their parametrisation such as volumetric capacity (m^3), infiltration surface (m^2) and infiltration rate (mm/m^2).

Runoff area (from map) can then be matched to the available storage volume and infiltration capacity. Based on statistical data on heavy rainfall events we can then estimate an infiltrated volume and spill over volume to the storm drain infrastructure.

The runoff of paved surfaces can be approached with the Curve Number Method (Soil Conservation Service Curve Number – SCS-CN), originally founded by the US Department of Agriculture in 1954 (Hawkins 2014). It has been used to predict direct runoff from rainfall events. Runoff varies according to the Curve Number (CN), which describes the antecedent potential water retention of a watershed. The CN value is related to hydrological conditions, land use and soil types. CN values for paved surfaces are derived for several types of urban areas. The SCN-CN method is because of its simplicity and limited required parameters widely used and implemented in hydrological models (e.g. SWAT) and storm water management models (Hawkins 2014; Jiao et al. 2015; Li et al. 2018; WUR n.d.).

We also will incorporate green roofs as an EbA measure. Green roofs should be combined with infiltration infrastructure. They act the same as infiltration systems, but their volumetric storage capacity is realised within the green roof. This allows to have smaller dimensions for the infiltration infrastructure, which is interesting when space is limited and/or soils are less permeable. The roof water storage capacity is determined by substrate type and thickness. The latter also determines the water holding capacity. In general, green roofs will have higher interception and evapotranspiration losses.

2.1.3. Infiltration and retention in soils

Literature

Soil degradation is important issue that has a strong influence on water regulation aspects such as runoff, erosion, soil water holding capacity and groundwater recharge. It will alter physical properties such as bulk density and porosity which on their turn will affect chemical soil properties, soil fauna and plant growth and diversity. Quantitative relationships are however hard to find in literature. Obviously, soil properties can be immensely diverse, and it is difficult to quantify cause-effect relationships without long-term studies. Many studies are conducted from an agricultural land management perspective, only recently there has been more attention for an environmental perspective to land management. We here refer specifically to degradation of soils under agricultural management.

Soil bulk density and water infiltration rate are often used as indicators for soil compaction, because the porosity is decreased. A reduced porosity does not only affect water infiltration, it has also an influence on the nitrogen cycle. Compaction causes an increase of denitrification as a result of an increase in water content. Moreover, anaerobic soil conditions are favoured which leads to a higher production of CH_4 and increases the emissions of this greenhouse gas (Nawaz et al. 2013).

Basche and DeLonge (2019) evaluated in a meta-review of soil management options the effects of five main soil management practices (no-till, cover crops, crop rotation, introducing perennials, crop and livestock systems) on infiltration rates compared to conventional practices (Figure 25). Cover crops were shown to increase infiltration compared to conventional tillage by 34.8% on average, independent of aridity and rainfall regime, but they seemed to perform better on coarser soil. Their effect also increased significantly if in place for more than four years (Figure 26). They also found that the overall effect of no-till was not significant, but in combination with residue retention, higher increases in infiltration were observed (Figure 27). In general, tillage decreases soil porosity, lowers the hydraulic conductivity and decreases the bulk density of soils which compacts the soil beneath (Gómez et al. 1999).

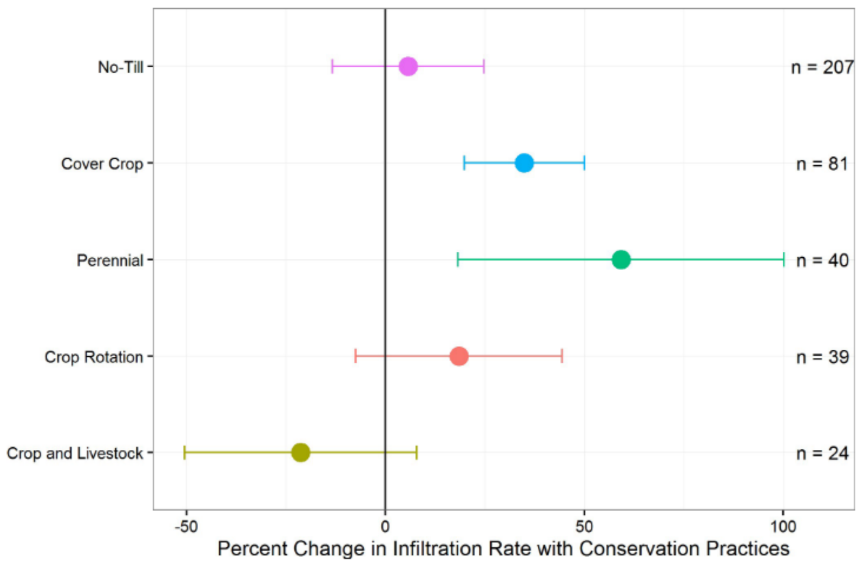

Figure 25: Change in infiltration rate for five agricultural practices compared to conventional conservation practices. In general, it was found that the overall effect of no till on infiltration rate was not significant compared to conventional practices. Introducing perennials and cover crops have the most positive impact on infiltration rates (Basche and DeLonge 2019).

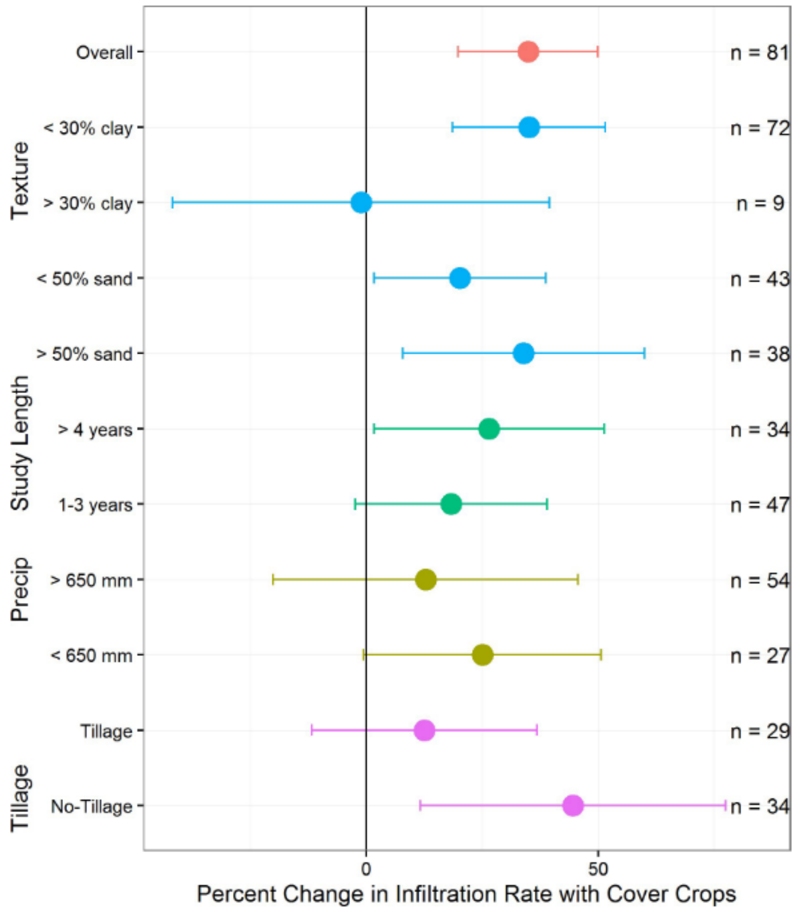

Figure 26: Change in infiltration rate for subsets of cover crops experiments (Basche and DeLonge 2019). These subsets include the use of additional agricultural practises (such as residue retention, cover crops) and environmental factors (such as soil texture variables, precipitation and study length). No till in combination with residue retention leads to increased infiltration rates. Cover crops were shown to increase infiltration compared to conventional tillage by 34.8% on average, independent of aridity and rainfall regime, but they seemed to perform better on coarser soil. Their effect also increased significantly if in place for more than four years (Basche and DeLonge 2019).

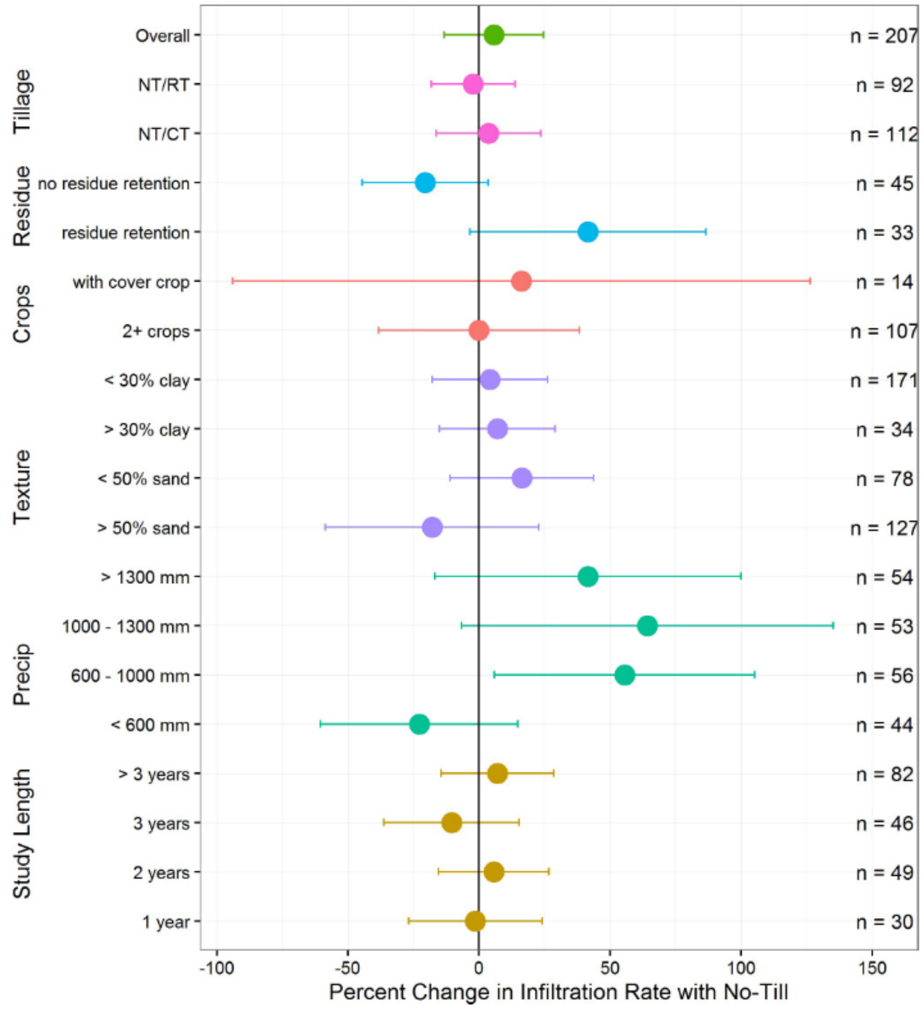

Figure 27: Change in infiltration rate for subsets of no-till experiments. These subsets include the use of additional agricultural practises (such as residue retention, cover crops) and environmental factors (such as soil texture variables, precipitation and study length). No till in combination with residue retention leads to increased infiltration rates (Basche and DeLonge 2019).

SOM plays also an important role in nutrient retention and provision to plants as well as soil particle aggregation, and so influences aeration, structure, drainage and other functions. SOM can hold up to 20 times its weight in water and makes soils more drought resistant (Dick and Gregorich 2004; FAO 2005).

Effects on soil water holding capacity

The soil water holding capacity is to a large extent determined by soil texture (Table 9). Heavy soils naturally have a high water holding capacity, but also a low water permeability, making them vulnerable to erosion and runoff. Light soils have a low water holding capacity and a higher water permeability, making them more vulnerable to desiccation. Therefore, objectives to improve soil structure can be different. For heavy soils, improving soil structure is especially important to maintain and restore macro pores, decreasing risk for runoff and erosion. In general, these heavy soils dry out very slowly, so there is less drought vulnerability, even without targeted management. For lighter soils, soil structure is equally important. Soil organic matter improves water holding capacity of sandy soils. Even on sandy soils, the topsoil is usually fine textured due to land tillage, which makes the topsoil vulnerable to disintegration and clogging of macropores.

Table 9: Soil moisture ranges for different soil types with indication of soil moisture content according to the textural category (Θ_s) and soil moisture content at field capacity (Θ_{FC}) (Houšková 2016).

SOIL TEXTURAL CATEGORY	SOIL HORIZON (m)	θ_s	θ_{FC}	PROPER SOIL MOISTURE for cultivation ($0.9\ \theta_{FC}$)
		(% of volume)		
Light soils	0 - 0.30	43.93	28.11	25.30
	0.31-0.80	44.26	27.71	24.94
	0.81-1.10	42.95	28.02	25.22
Medium heavy soils	0 - 0.30	45.35	34.09	30.68
	0.31-0.80	41.98	33.74	30.37
	0.81-1.10	40.47	33.25	29.93
Heavy soils	0 - 0.30	45.99	37.52	33.77
	0.31-0.80	43.83	36.91	33.22
	0.81-1.10	42.13	36.52	32.87
Very heavy soils	0 - 0.30	49.23	40.15	36.14
	0.31-0.80	45.15	40.9	36.81
	0.81-1.10	45.51	40.09	36.08
Clay	0 - 0.30	50.17	43.5	39.15
	0.31-0.80	50.06	45.44	40.90
	0.81-1.10	50.54	47.87	43.08

A high water holding capacity does not imply that there is a large buffer storage. Soil wetting and drying phases are simply much slower for heavy soils. Sandy soils have a low matric potential (Figure 28). They can quickly release water from their pores to provide groundwater recharge and base flow. Silt and clay soils have a high water holding capacity (Figure 28). This is positive for the vegetation but provides less base flow and aquifer recharge. Because PROWATER focusses mainly on improving base flow and deep infiltration, the soil water holding capacity of the parent material is of importance. We aim to promote a fast and thorough (re)wetting of the soil during precipitation events. This should reduce runoff. But evidently there are differences in the drainage phase too.

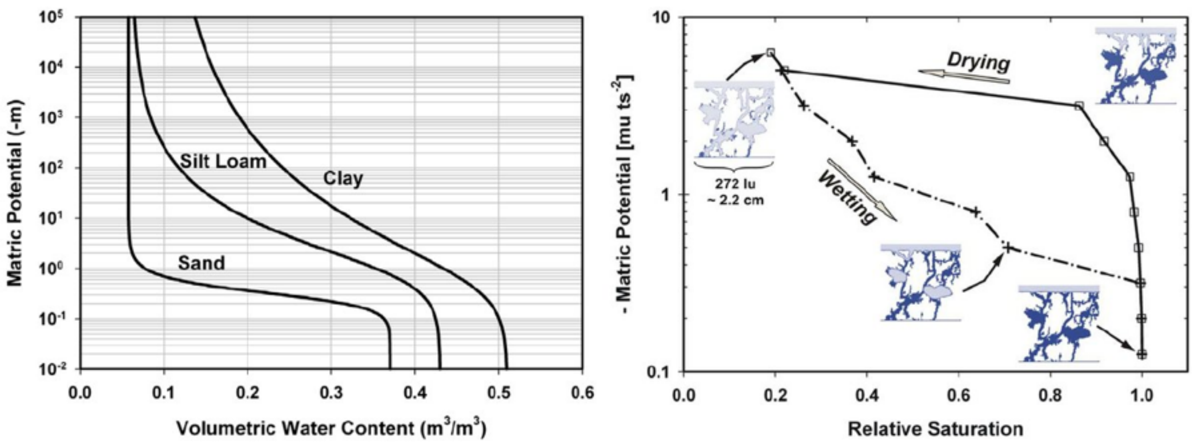

Figure 28: Soil water content curves for soils of different textures which illustrate the effect of porosity (saturated water content) and the varied slopes of the relationships resulting from different pore size distributions (Tuller and Or 2003).

Table 10 shows that the relative impact of compaction on soil water holding capacity is higher for light textured soils. Given that sandy soils already have a lower soil water holding capacity, the importance is even more significant. This is because sandy soils compared to clay soils will dry faster as explained above. Compaction leads to a higher bulk density, thus decreasing the amount of pore space for water, and therefore sandy soils dry out even faster.

Table 10: Effect of soil compaction on soil water holding capacity for various soil types. Soil density ρ_d, soil moisture content Θ_p and water holding capacity (RVK) are compared for compacted and non-compacted soils for various soil types (Houšková 2016).

Soil type	Compacted soil		Non-compacted soil		Compacted soils	Non-compacted soil
	ρ_d (g.cm^{-3})	θ_P (% obj.)	ρ_d (g.cm^{-3})	θ_P (% obj.)	RVK (m^3.ha^{-1})	RVK (m^3.ha^{-1})
Clay	1,5	43	1,35	49	2 322	2 646
Clayey soil	1,6	40	1,4	47	2 160	2 538
Loamy soil	1,65	38	1,45	45	2 052	2 430
Sandy-loamy soil	1,75	34	1,55	42	1 836	2 268
Loamy-sandy soil	1,8	32	1,6	40	1 728	2 160
Sandy soil	1,9	28	1,7	38	1 512	2 052

Improving infiltration and water holding capacity through improved soil quality will directly benefit farmers, but also enable a faster rewetting of soils. A more erratic weather pattern will result in both extreme wet and dry periods. Compacted soils will have more frequent and severe periods of stagnant water from summer storms and are slower in their rewetting phase after droughts. Non-reversing tillage, compost amendments and crop residues will improve the infiltration and water holding capacity of the topsoil, but is unlikely to improve infiltration when the subsoil is compacted (tillage pan).

<u>Effects on maximal soil water infiltration capacity</u>

The maximal soil water infiltration capacity determines the ratio between infiltration and runoff for certain rainfall intensities. Again, this depends mainly on the soil type. The infiltration capacity will be low for clay, but still can be improved significantly by SOM. So therefore, important focus on the impact of compaction, bioturbation, macropores and SOM.

The maximal soil infiltration capacity (mm/h) of the topsoil in unsaturated conditions is important to assess the runoff from unvegetated soils. There is evidently a strong link to the prevention of soil erosion. In the soil erosion model, there are two parameters that can incorporate the impact of management on erosion, namely the erosion control practice factor P (unitless) and the factor for topographic soil erodibility, based on slope percentage and slope length LS (unitless).

LS is mainly used to incorporate effects of soil vegetation cover and physical barriers on slope length. Basically, measures that capture eroded particles on their downward journey after splash erosion occurred.

P can be used to incorporate a decrease of soil erodibility through improved soil infiltration capacity and reduction of splash erosion by SOM.

Impact of soil quality will only be applied on exposed agricultural soils (non-permanent vegetation). Again, we use the base situation of croplands with topsoil compaction and low SOM. This allows to correct this in a positive way by making use of management parameters (intercropping, crop residues, compost amendments, deep ripping, wheel tracking).

Effects on maximal soil water storage capacity

EbA measures, such as non-reversing tillage, compost amendments, catch crops and crop residues, can be implemented to restore and protect soil health with an aim to support infiltration and retention of water and practices relating to soil health. These EbA practices will improve the infiltration and water holding capacity of the topsoil, but is unlikely to improve deep infiltration when the subsoil is already compacted (tillage pan). In many cases infiltration capacity is also hampered by deep soil compaction. When the soil permeability of deeper layers is lower than topsoil infiltration capacity, saturation will occur, which results in runoff.

Current representation within ECOPLAN-SE

There is currently no representation of soil quality effects on infiltration rates. Annual infiltration is currently reduced for soils that are expected to have shallow groundwater levels. When soil depth to the mean highest groundwater level (MHGW) is less than 75 cm below field level (FL), we reduce annual infiltration gradually. For a sandy soil, we reduce the maximal annual infiltration of 450 mm/y with the following formula.

$$Infiltration = 450 - (450 * (75 - MGHW))/(75))$$

Example: If the groundwater level is 30 cm below field level, then infiltration is calculated as follows:

Infiltration = *450 – (450* ((75-30)/75))*
*450 – (450 * 0.6)*
450 – 270 = 180 mm/y.

The runoff generation on groundwater saturated soils is not explicitly quantified, because a large part of the infiltration loss is evapotranspiration. Actual runoff will only be generated during winter and when groundwater levels are the highest.

The ECOPLAN-SE erosion model predicts soil erosion and where water flows are expected to converge on the field scale, but does not quantify the associated water volumes and does not interact with the infiltration model. The missed infiltration quantities from soil runoff events were considered to be negligible compared to total annual infiltration. Field observations show that runoff from agricultural fields occurs frequently, also in flatter areas and also after long periods of precipitation.

Ambitions for PROWATER

By applying measures on agricultural land that improve soil cover and soil structure, we can avoid runoff and promote infiltration. This is primarily targeting the improvement of the topsoil quality by reducing tillage and increasing SOM. This will especially increase the maximal infiltration capacity of the topsoil.

Remediation of deep soil compaction is not targeted within PROWATER. Yet we do want to include this aspect in the modelling. To account for deep soil compaction and its remediation, we can assume a limited infiltration rate (e.g. 2-5 mm/day when compacted) for a deep (20-40 cm) compacted layer. These assumptions allow to calculate the potential accumulation of water from topsoil infiltration. This enables us to simulate a saturation (water logging) of the topsoil and associated runoff. We will to model in a simplistic way how the soil profile is saturated when precipitation surplus exceeds the deep drainage capacity. This approach can also be used for areas with shallow groundwater levels as these act similar to slowly permeable layers.

Yet not all runoff can be avoided, and runoff generated by topsoil saturation on compacted subsoils after persistent rainfall will be difficult to avoid. By applying measures on the micro-scale runoff collection zones, we can slow down the hydrological pathways. Firstly, there is a water volume that is retained within runoff collection ponds and ditches, secondly there is a deferred deep infiltration over time and thirdly, there is a volume that could not be stored and directly drained to the river network. The collection and storage of runoff water in micro-scale wetlands is discussed in 2.1.4.

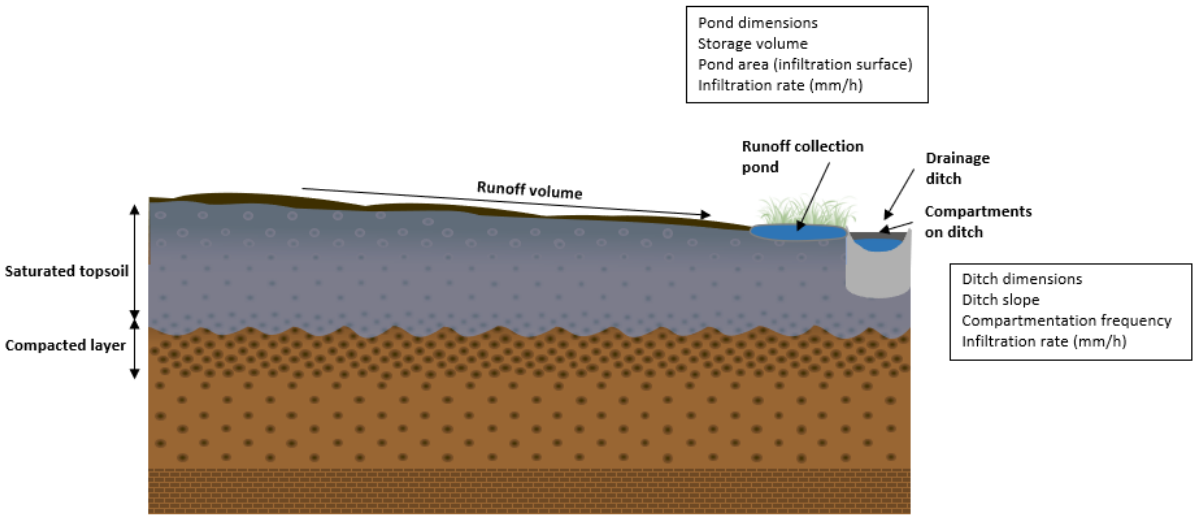

Figure 29: Remediation of deep soil compaction, runoff due to topsoil saturation and associated measures, such as runoff collection ponds.

2.1.4. Water retention and storage in wetlands and surface water

Literature

Wetlands play an important role in hydrology and offer numerous environmental functions (Bullock and Acreman 2003; Kadykalo and Findlay 2016). Their functioning is strongly dependent on the hydrogeomorphic setting. Brinson (1993) describes wetlands based on hydrodynamic differences within different geomorphic settings. This hydrogeomorphic classification contains three core components (Mitsch and Gosselink 2007) as mentioned below. A more detailed description of the different wetland forms according to the hydrogeomorphic classification can be found in Brinson (1993).

- The geomorphic setting describes the topographic location of a wetland in the surrounding area. This can be either depressional, riverine, fringe or extensive peatlands.
- The contribution of three water sources (groundwater, precipitation and surface flow) as shown in Figure 30.
- Hydrodynamics refer to the water movement within a wetland. Brinson (1993) distinguishes vertical fluctuation, unidirectional flow and bidirectional flow. The first is typical for depressional wetlands. Riverine wetlands have more unidirectional surface or near-surface flow while bidirectional flows mostly occur in fringe wetlands.

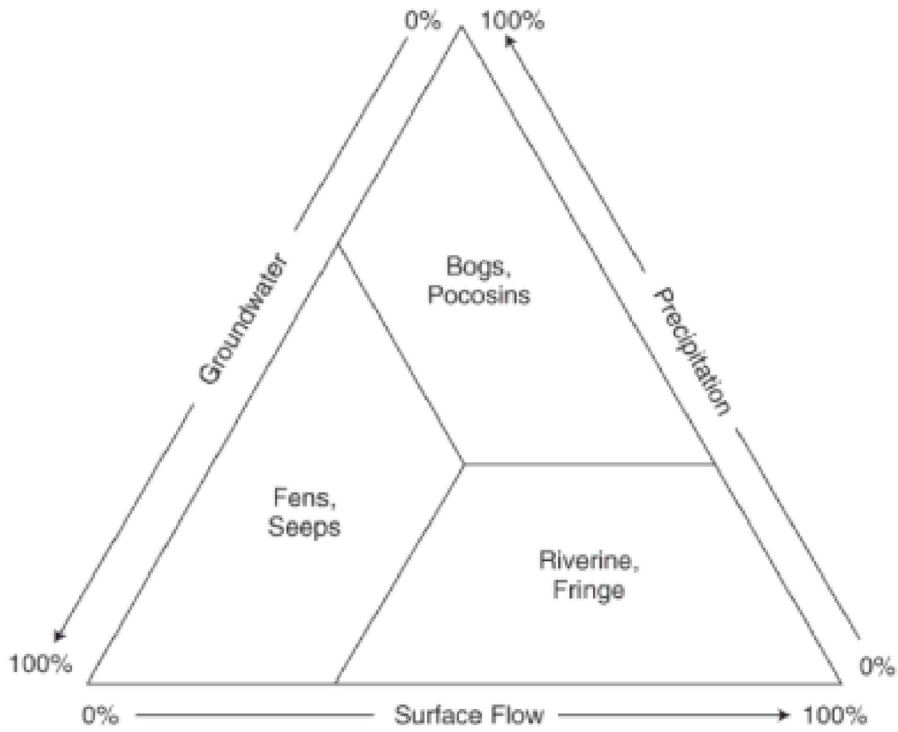

Figure 30: Relative contribution of three water sources (Mitsch and Gosselink 2007, after Brinson 1987)

Lane et al. (2018) describe the source function of hydrologically connected wetlands to downstream waters. Non-floodplain wetlands are a source when total water input (e.g. surface or groundwater inflow) minus output (e.g. evapotranspiration, groundwater recharge) exceeds the storage capacity of the basin. Connections are made through surface, subsurface or groundwater flow and vary in frequency, timing, duration and magnitude.

Non-floodplain wetlands also act as hydrological sinks when the storage capacity passes net inflows such as surface water and groundwater. The hydrological sink function at the watershed scale is maximised in the case of the combination of runoff, low discharge rates and high surface storage capacity. Evapotranspiration losses can be large, while streamflow and peak flows are reduced as a result of the water storage capacity of wetlands (Lane et al. 2018).

Figure 31 shows wetland connectivity in relation to the landscape position of wetlands (Fan and Miguez-Macho 2011). Upstream wetlands are connected through several hydrological paths to the downstream waters and vary in time and space. Hydrological flow paths vary from surface flow paths (runoff small streams) to subsurface flow paths and the position in the landscape determines the travel time in the watershed (Cohen et al. 2016). Figure 32 shows the cumulative effect of many upstream (isolated) wetlands on the watershed travel time (Rains et al. 2016).

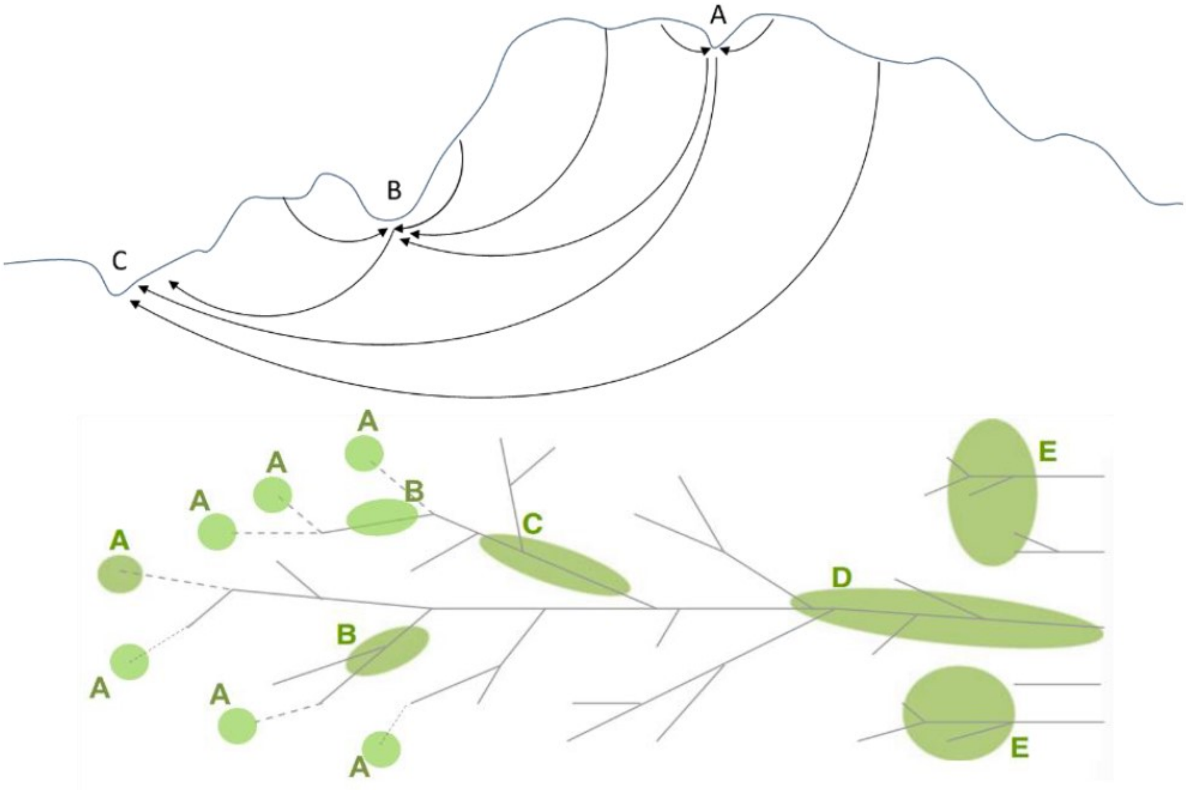

Figure 31: Wetlands according to their topographical position which emerges infiltration and seepage patterns in the landscape (adapted from Fan and Miguez-Macho, 2011). A-type of wetlands are upstream landscape depressions which are temporally wet. B-type wetlands are headwater wetlands which form source areas for small streams. C-type wetlands are valley bottom wetlands with permanent seepage. D-type wetlands are flood inundation areas related to larger rivers and E-type wetlands are under influence of tidal regimes. With PROWATER we mainly focus on the A-, B- and C-type wetlands.

- Flood plains and valley bottom wetlands (C- and D-type wetland in Figure 31, green in Figure 32) linked via (perennial) surface flow paths give rise to dynamic storage during high flows. They experience the shortest travel time.
- Hydrological connectivity via intermittent fill-and-spill dynamics is typical for vernal pools characterised by episodic and transient surface flow paths (yellow in Figure 32).
- Upstream depressional wetlands, connected via subsurface flow paths, have the largest travel time (A-type wetland in Figure 31, red in Figure 32). It has previously been observed that they restrain peak flow and control base flow. Water storage in this type of wetlands has a substantial effect on downstream flow and promotes groundwater recharge.

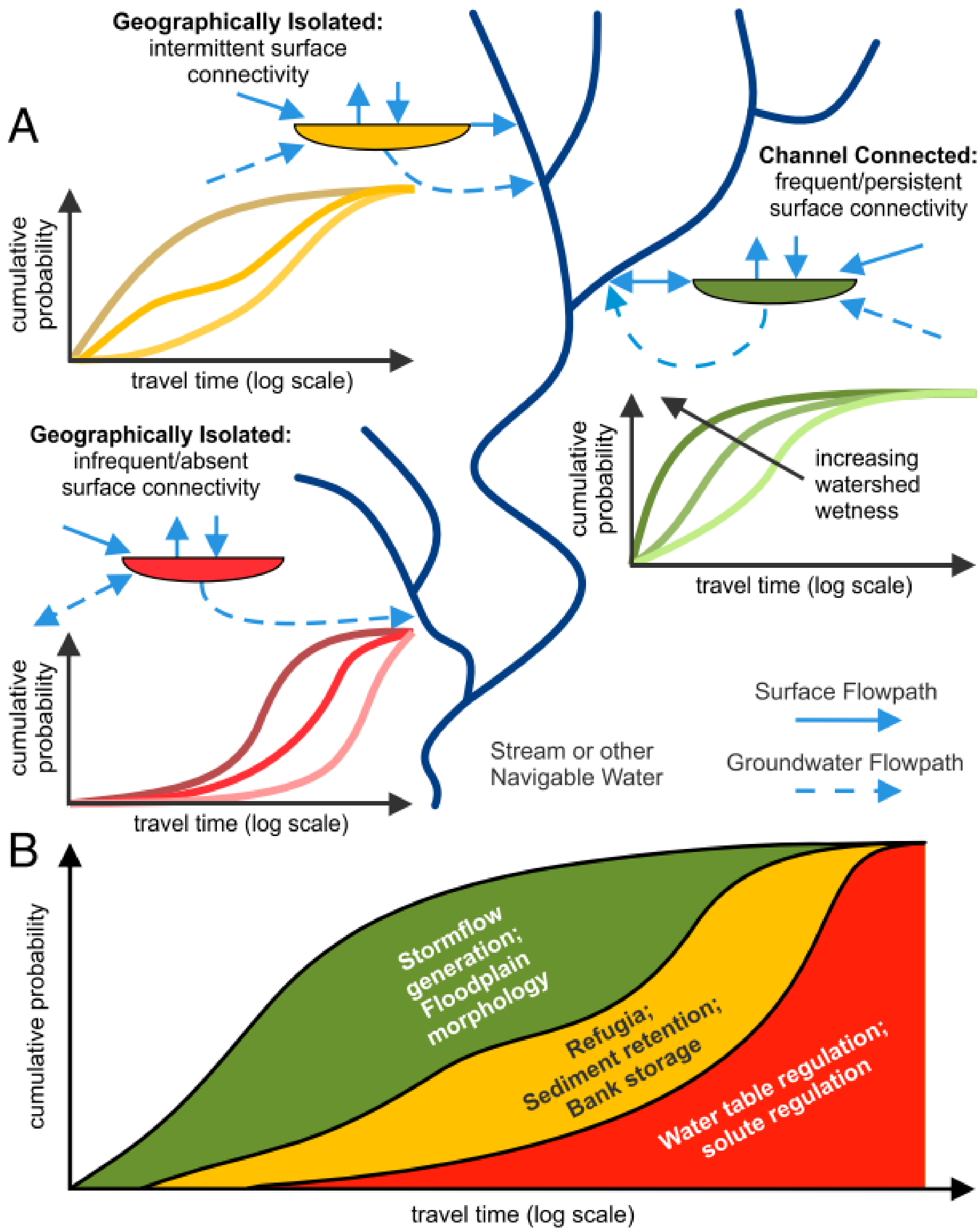

Figure 32: (A) Hydrological connectivity between upstream and downstream wetlands. Cumulative travel times are shown for the several wetland types according to their position on the stream network. (B) Cumulative travel time as a result of convolution of the individual travel responses of the wetland along the hydrological connectivity spectrum (Cohen et al. 2016).

This hydrological connectivity between wetlands and downstream waters has been altering for decades which affects the hydrological response. Drainage networks significantly reduce the water storage capacity of wetlands (Jones et al. 2017; Rains et al. 2016).

Wetland storage capacity is defined by Jones et al. (2017) as ‘the maximum surface water volume each depressional basin can store without spilling to downgradient waters’. Several methods currently exist for the measurement of water storage capacity. These include techniques using topographical data, empirical relationships and contour-based approaches (Jones et al. 2017). Studies have modelled storage volumes applying relationships between volume and wetland surface area (Gleason et al. 2007). For instance, Lane and D’Amico (2010) made use of LiDAR to estimate the potential water storage of geographically isolated wetlands. LiDAR-derived measures were compared with volumes calculated from equations related to different geology formations. It follows that basin morphology must be included in order to improve accuracy. Jones et al. (2017) developed a novel raster-based approach to estimate the restorable wetland water storage of depressional basins. First, they used LiDAR-derived topographic data to delineate depressional basins. Second, they generated relationships between inundation depth, area and storage volume. Finally, they calculated the spill threshold for each depression to estimate the current and potential storage capacity. The latter can be achieved via, for example, ditch filling. Figure 33 demonstrates loss of wetlands storage following drainage (orange line).

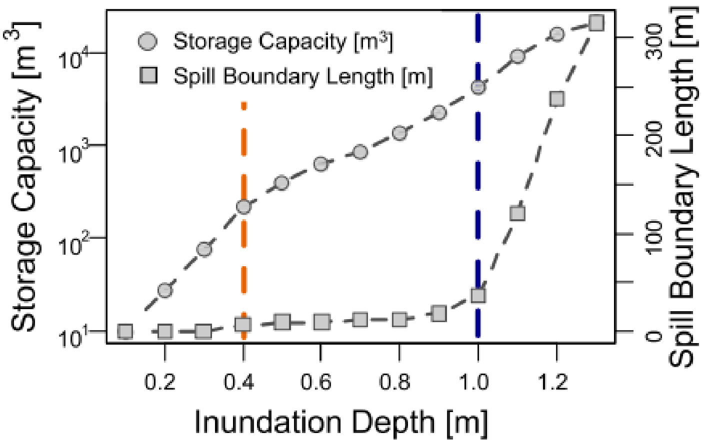

Figure 33: Storage capacity and spill boundary length related to inundation depth. The orange line corresponds to estimated contemporary and the blue line to potential storage capacity. The spill boundary length is equal to the length of the depression boundary that is inundated (Jones et al. 2017).

Water retention and storage in wetlands and surface waters refers to the water in the subsoil that is available to provide base flow and/or result in deferred groundwater recharge. It is different from the runoff collection and retention, because these processes occur on the landscape scale and mostly involve larger, natural depressions within the landscape. These landscape depressions are temporarily to permanently waterlogged. The timing, frequency, and duration of inundation or waterlogged conditions can be collectively considered as the “hydroperiod” of wetlands. Hydroperiod is an overarching metric for hydrological integrity, encompassing concepts such as flood-pulse, ecological flows and so on. Hydroperiod is crucial to all types of wetlands, including rainfed, groundwater fed and surface water systems (incl. riverbed and floodplains). Changes in hydroperiod lead to changes in ecological structures and patterns. This affects the delivery of wetland ecosystem services, for example climate regulation, fish nursery and water purification (Brooks 2000; Calhoun et al. 2017; Hamilton 2009; Riley et al. 2017). Figure 34 shows the hydroperiod in a hydrograph (wetland stage in function of time).

Small scale upstream wetlands or Upstream Depressional Wetlands (UDWs), also called depressional, geographically isolated of non-floodplain wetlands (wetland types A and B in Figure 31, red and yellow in Figure 32) are naturally characterized by a high fluctuation in water levels due to their topographical position. They have a hydroperiod with a short lag time, high frequency and low amplitude. This creates possibilities for deferred infiltration which recharges groundwater reserves and increases base flow during subsequent periods of drought. However, following Decleer et al. (2016) most of these wetlands are currently drained for agricultural purposes (Infascelli et al. 2013; Merot et al. 2006; Montreuil and Merot 2006). The peaks in water level are more flattened in the case of drainage. As a result, the (change in) hydroperiod gives an indication of how much water can be lost through drainage. Downstream valley bottom wetlands diminish peak flows by flooding and reduce base flow during periods of droughts (Staes et al. 2009).

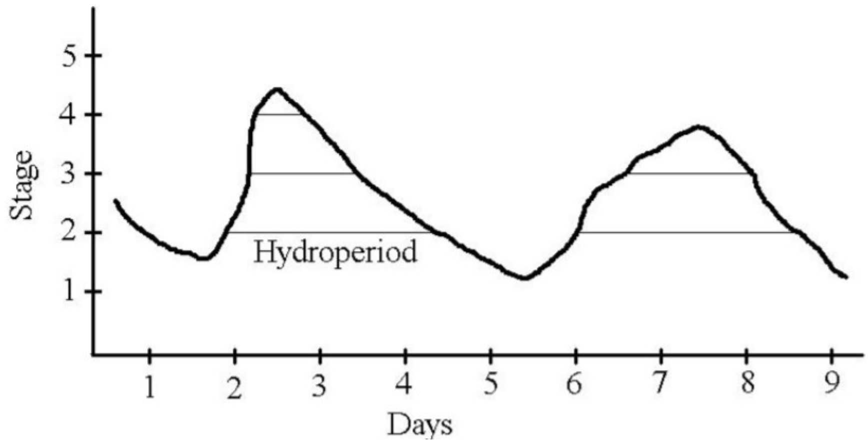

Figure 34: Hydrograph showing the water level in function of time and the hydroperiod (U.S. EPA 2008).

Current representation within ECOPLAN-SE

The current version of the ECOPLAN-SE (Vrebos et al. 2017) considers water retention as the volume of water (m³) that is present up to one meter below ground. This amount of water contributes to the feeding of watercourses and deep infiltration during drought periods. The ECOPLAN-SE calculates both seasonal and permanent water retention. It refers to the capacity to deliver water to surface waters and groundwater aquifers in case of droughts, which is crucial for supporting ecosystems and river base flow. Seasonal retention adverts to the buffer capacity between the mean highest and mean lowest groundwater level. This equals the difference between spring and autumn water levels. Seasonal retention is important in the context of flood prevention and groundwater recharge. Permanent retention adverts to the total drainable water content in case of extreme droughts. The assumption is made that all uncategorized watercourses have a draining function. This is however a premature premise as these watercourses could also have an infiltration function due to the collection of water from paved surfaces. No distinction can be made between draining and infiltrating watercourses based on the available data in the Flemish Region.

To calculate drainage, we take different factors are taken into account. We first determine a preferred drainage depth based on soil cover and land use. Secondly, we estimate the drainage ditch density from a digital elevation map and data on the hydrological network. The impact on the mean highest groundwater level (MHG) and mean lowest groundwater level (MLG) depends on the distance to the drainage ditch (density of ditches), the soil type (permeability) and the preferred drainage depth.

Finally, indicators are calculated that express the relative loss of retention, compared to an undrained situation.

Ambitions for PROWATER

UDWs can be identified by means of the Topographic Position Index (TPI). This is a relative topographical index that shows the difference between a central point and the mean elevation in the neighbourhood around the point (Weiss 2001). It takes into account the variability or range in elevation in the neighbourhood around the point, by normalizing the elevation to the variability or the range. A TPI signature for a wetland can be generated from the combination of the TPI on different spatial scales. The course of the water level (and thus the hydroperiods) can be simulated by means of the combination of all the TPI signatures for an UDW.

This only considers the available water in saturated layers of the subsoil. We limit this to one meter below average field level, because that encompasses the volume that can potentially be drained. Free draining soils will have no barriers for deep infiltration once the soil profile is saturated. But many soils have subsoil barriers that impede deep infiltration.

The presence of natural poorly permeable layers can result in subsurface water flows. We assume that these areas are depicted by the spatial prioritisation tool as temporary wetlands. A soil depth layer, soil vertical and horizontal permeability and topographic position index can be used to estimate subsurface flows and potential local water saturation. Retaining water within temporary wetlands can result in reduced peak flow, increased base flow and deferred infiltration. Where water flows are expected to converge on the landscape scale, we can include the management of drainage systems and the concept of deferred infiltration for temporary wetlands. When there is no drainage, the waterlogging will persist and deferred infiltration may still take place. When drainage is assumed, we assume the built up water volume to be partly lost.

The actual and potential water storage capacity of a wetland can be calculated in GIS based on the Digital Elevation Model (DEM). The volume can be derived from the mean elevation of the depression boundary and is calculated for with and without drainage. It is also possible to calculate the water storage distribution of a wetland.

The storage/retention/drainage module will allow to define properties over time and setting basic management rules. As a base line, we assume that intensive agriculture and urban land uses have deep drainage.

2.1.5. Deep infiltration and aquifer recharge

The tool starts from the maximum infiltration potential of the soil, which is determined by the groundwater depth (a high groundwater level will limit infiltration) and soil texture (heavy soils, such as clay soils will have a much lower infiltration rate). Then the actual infiltration is limited by soil compaction and interception. The first is with regard to paved surfaces which are not connected to drainage and sewerage system. The latter is dependent on land use.

2.2. Erosion prevention

Literature review

The ecosystem service of erosion prevention is typically characterised by a spatial mismatch between supply and demand. The service is supplied in an upstream area where changes in vegetation and/or land use may reduce soil loss, but the beneficiaries are both the owners of the land where soil loss is prevented as well as downstream land owners where sediment-loaded runoff passes (mudflow) and where sediment is being deposited (Figure 35). When integrating erosion control in a payment for ecosystem services scheme it is crucial to map both the suppliers and the beneficiaries. Different methodologies will be necessary to map sediment source, flow and sink (Bagstad et al. 2011), which can be combined and coupled into a single model (e.g. WATEM-SEDEM).

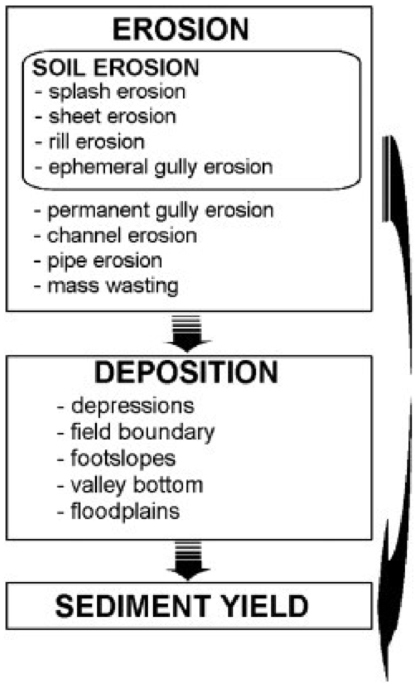

Figure 35: Erosion-sedimentation processes relevant in assessing erosion control as an ecosystem service (de Vente et al. 2008)

A range of models is available to quantify some or multiple erosion-sedimentation processes (de Vente et al. 2013). In this review we only include quantitative models that (1) can be used on medium to large spatial scales (excluding models with high data requirements such as physics-based models because of limited data availability in some areas); (2) are suitable to evaluate scenarios of land use change and erosion control measures (see review by de Vente et al. 2013) and (3) include information both on the sediment source area as well as on the sediment deposition area.

One of the most commonly used approaches to quantify the capacity of (natural) vegetation to reduce erosion is through calculation of the Revised Universal Soil Loss Equation (RUSLE) by Renard et al. (1997) (Xu et al. 2020).

$$Erosion\left(\frac{ton}{ha.y}\right) = C \times K \times R \times P \times LS$$

With:

- C: crop management factor (unitless)
- K: soil erodibility in ton.hr/(MJ.mm)
- R: rainfall erosivity in MJ.mm/(ha.yr)
- P: erosion control practice factor (unitless)
- LS: factor for topographic soil erodibility, based on slope percentage and slope length (unitless)

The equation combines data on topography, soil texture, vegetation type and precipitation to quantify the volume of sediment that is lost per year due to rill and inter-rill erosion caused by overland flow of rainwater. Erosion prevention can then be calculated by comparing the volume of eroded sediment in two scenarios, for example a scenario with natural vegetation (reduced erosion) and a scenario with intensive agriculture or bare soil (high erosion). The RUSLE equation is to be applied with rasters, preferably with a resolution close to 20x20 m as the equation was developed based on this resolution and it delivers the most accurate results. One of the main limitations of the RUSLE equation is that it only takes into account rill and inter-rill erosion and does not account for other sources of sediment such as gully erosion, streambank erosion and mass erosion (de Vente et al. 2013; Sharp et al. 2018).

However, it has an important advantage that it is low in data requirements, facilitating its application on larger spatial scales. On top, its universal usage allows to compare results among different regions and studies. It also constitutes the basis for the erosion models integrated in the ecosystem services toolboxes InVEST and ARIES. The latter complements RUSLE with regional Bayesian erosion models, including the method of Egoh et al. (2008) for steeper slopes or areas where RUSLE is known to be inadequate (Bagstad et al. 2011). Other spatially explicit erosion models that use RUSLE or comparable equations in combination with sediment flow and sink models are WATEM-SEDEM and SPADS (see further for a brief discussion of the models).

On very large spatial scales (regional or national), it becomes more challenging to use RUSLE as it should ideally be applied on a resolution of 20x20 m and is calibrated for this resolution. Alternative methods for large-scale erosion assessments used in ecosystem services studies are mostly based on vegetation. For example Burkhard et al. (2012) and Kandziora et al. (2013) use percentage vegetation cover as indicator for soil erosion potential. This gives an indication of erosion susceptibility, but it does not allow to accurately quantify soil loss. To quantify actual erosion reduction it is necessary to have information on soil type (texture) and topography. Zhongming et al. (2010) take this a step further by taking into account the stratification of the vegetation (layered vegetation). This stratified vegetation index can be used as a stand-alone indicator for potential erosion, or in combination with erosion models such as RUSLE to have a better representation of the capacity of the vegetation to reduce erosion when quantifying actual erosion. Egoh et al. (2008) use a different approach by combining estimates of soil erosion potential with expert-based assessments of the capacity of the vegetation or litter layer to reduce erosion.

After applying the RUSLE formula, the erosion module of InVEST quantifies the amount of sediment retained by vegetation and topographic features (sediment delivery ratio), based on the work of Borselli et al. (2008). The proportion of soil loss actually reaching the stream is assessed based on hydrological connectivity of the pixel where erosion occurs (modelled with RUSLE) with the stream. This is translated into a weighing factor by which the RUSLE equation is multiplied. The model also quantifies the amount of sediment that is deposited on the landscape downstream from the source but that does not reach the stream (mud deposited where mudflow passes), by multiplying RUSLE with (1 - weighing factor).

ARIES applies different models to map sediment flow and sink. The sink model takes into account stream gradient, floodplain tree canopy cover, floodplain width and dams to quantify and map the amount of sediment deposited in a floodplain or reservoir (tons/y). The modelling is based on Bayesian principles where the different parameters are discretized into classes and a probability is assigned to the amount of sediment deposition that is attributed to each combination of classes (e.g. greatest deposition in low-gradient streams with wide floodplains and high levels of tree canopy cover). Beneficiaries of avoided erosion are mapped by simply overlaying their location or terrain (e.g. reservoirs, drinking water intakes) with the results of the sediment sink or source calculations. ARIES also allows to take into account sedimentation of eroded soil as a benefit. In that case erosion (and subsequent sedimentation) becomes a provisioning ecosystem service. Beneficiaries are mapped in a similar way as for the regulating service of erosion prevention (by overlaying the terrains of the sediment beneficiaries with the results of the sink models). An additional model can be applied to identify sediment flow and to map areas where sediment is deposited before reaching a stream. This is based on a simple hydrologic model that derives flow direction from hydrography (stream network) and elevation data, taking into account floodplains extent, levees and dams. ARIES does not include an economic valuation of sediment regulation.

WATEM-SEDEM (Van Oost et al. 2000) combines RUSLE with a spatially distributed transport capacity to route sediment across the landscape and to model sediment deposition. Additional parameters that are required besides the parameters of RUSLE are land use or land cover and soil conservation practices. SPADS (de Vente et al. 2008) is based on similar parameters as RUSLE (mean monthly rainfall depth or intensity or R factor, Digital Elevation Model, land use or land cover or C factor, and soil type, texture or soil erodibility estimates). Based on these parameters and derivatives slope, gully density and inverse distance from river stream, SPADS calculates an expert-based index and relates this to mean annual sediment yield through regression. The result is an index map rather than a quantification of the annual amount of erosion and deposition (de Vente et al. 2008).

Other models to assess sediment yield are BQART, WBMsed and Pelletier's model (de Vente et al. 2013). The BQART-model (Syvitski and Kettner 2007) quantifies suspended sediment load to rivers on large spatial scales using parameters catchment area, large-scale relief, mean annual temperature, river discharge, lithology, ice cover and human impacts, such as dams, land use and erosion control measures. It quantifies the amount of sediment load to major rivers but does not allow to make spatially explicit assessments on a pixel basis, making it less suitable for scenario analyses. The WBMsed-model (Cohen et al. 2013) couples BQART with a water balance model, allowing to perform a similar calculation as the BQART-model but on a pixel basis (each pixel is considered as an outlet of its upstream contributing area). The model of Pelletier (2012) is a spatially distributed non-linear regression model that quantifies suspended sediment yield at a global scale as a function of slope gradient, mean monthly rainfall, soil texture (fractions of clay, silt, sand and gravel), mean monthly Leaf Area Index (LAI) and a calibration factor. The model is rather data-intensive (29 parameters should be provided). However, none of these models allow to identify the sediment source area and hence to map suppliers of the erosion control service. The potential to couple these models with sediment source models deserves further investigation.

Generic ecosystem services models such as InVEST and ARIES do not make a link with economic value, for the economic value is said to be highly dependent of the particular application (who benefits?) and context (Sharp et al. 2018). Hence, economic valuation should be performed independently of these tools. Alam (2018) provides an overview of potential valuation techniques for erosion control for different beneficiaries, including: (1) avoided costs for agricultural conservation practices such as crop rotation; (2) costs to compensate for nutrient losses in agricultural production; (3) value of forgone yield (e.g. due to productivity loss associated with reduced nutrients); (4) dredging costs (waterways and retention basins); (5) equipment depreciation; (6) increased operating costs for drinking water companies (due to higher nutrient loads); (7) loss in drinking water withdrawal capacity due to sedimentation in watersheds; (8) avoided property damages caused by mud flows; (9) poor water quality and algal bloom damaging fisheries habitats. Costs are born as private costs (1-3), business costs (4-7) or societal costs (8-9).

Current representation within ECOPLAN-SE

In ECOPLAN-SE erosion prevention through water is calculated using the RUSLE equation (Vrebos et al. 2017).

- The C-factor is a measure of the effect of soil cover and cultivation on soil erosion. A C-factor is incorporated for every land use type and crop.
- The K-factor is a measure of the erodibility of the soil. A K-factor is linked to every soil texture class.

- A value of 880 MJ.mm/(ha.yr) is used as a mean for the Flemish Region for the rainfall erosivity factor R.
- The erosion control factor (P-factor) is currently not implemented.
- The LS factor in the RUSLE equation increases evenly with the slope length, following the principle that all sediment eroded upstream of a pixel will end up in the downstream pixel. A correction was applied to the LS-factor in ECOPLAN to account for upstream vegetation that reduces the amount of erosion reaching a downstream pixel. The correction factor is based on soil permeability and infiltrability associated with different vegetation types. Erosion reduction by management measures such as grass strips and levees, is also accounted for by modifying the LS-factor.

In the ECOPLAN project mud source areas are identified based on (1) the output of the RUSLE equation, (2) slope gradient, (3) valleys and open slopes derived from a Landform Classification Tool based on elevation data, and (4) connectivity between dry valley and a source of erosion (RUSLE output). Sediment flows are also identified using the Landform Classification Tool. The tool only identifies the source and flow areas but does not quantify the amount of soil that is deposited along the mud flow track.

2.3. Water purification (nitrogen removal)

Literature

Water purification is the ability of ecosystems to improve water quality. This includes chemicals, nutrients, sediments, contaminants and other substances in surface and groundwater. The water quality is of utmost importance for drinking water supply. Nutrients and other substances originated from decomposition, fertilizer application or atmospheric deposition are transported and end up in surface and groundwater (Brauman et al. 2007).

Different mechanisms performed by vegetation, soils, and microbes are responsible for removing pollutants from overland flow and groundwater (Brauman et al. 2007):

- physically trapping of water and sediments;
- lowering water speed to enhance infiltration;
- adherence to contaminants;
- biochemical transformation of nutrients and contaminants;
- absorption of water and nutrients from the root zone;
- stabilisation of eroding banks;
- dilution of contaminated water.

Contaminants are taken up by plants roots and microbial communities. Especially in wetlands the abundant macrophytes and microbes improve water quality by nutrient uptake and storage, sedimentation, adsorption, decomposition and denitrification (Blackwell and Pilgrim 2011; Brauman et al. 2007; Jordan et al. 2003; Moor et al. 2017; Verhoeven et al. 2006; Woltemade 2000). In oxygen-depleted conditions denitrification converts nitrate (NO_3) into N_2 emitted to the atmosphere. The duration of the interface and extent of interaction with contaminated surface water are decisive (Blackwell and Pilgrim 2011). In case of high nitrate loading, low pH or partial anoxia, also the greenhouse gas N_2O is formed, posing a trade-off (Blackwell and Pilgrim 2011; Moor et al. 2017; Verhoeven et al. 2006). Figure 36 shows different mechanisms of water purification in a wetland.

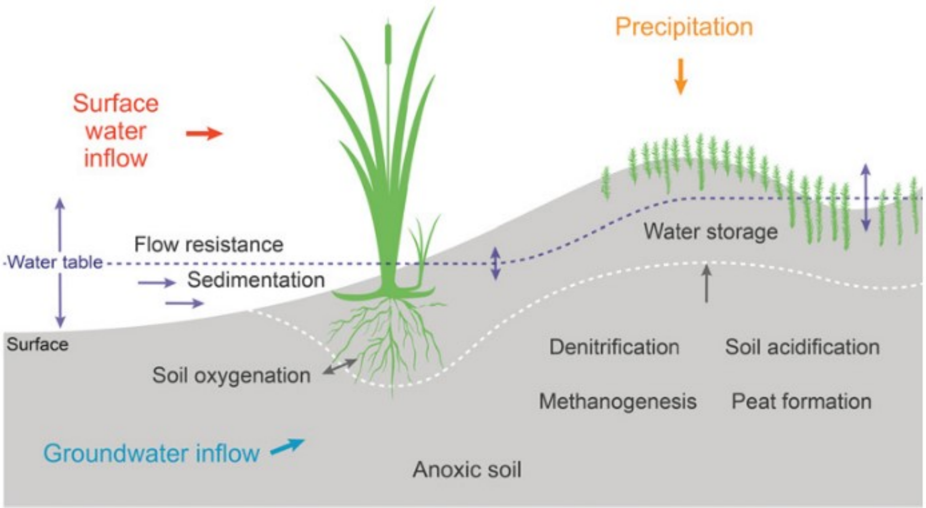

Figure 36: Different processes involved in water purification. The variation in water table is decisive for the occurrence of biogeochemical processes, such as denitrification and methanogenesis (Moor et al. 2017).

These biogeochemical processes in wetlands depend on water residence time (hydroperiod) and hydrological connectivity between nutrient sources and sinks (Calhoun et al. 2017; Marton et al. 2015; Powers et al. 2012). UDWs are biogeochemical hotspots because they are the primary receiver of nutrients and pollutants and prevent them from downstream transport (Cohen et al. 2016). The wetland edge is the place were biogeochemical processes tend to occur (Calhoun et al. 2017; Cohen et al. 2016).

- The biogeochemical activity is directly proportional to the perimeter/area-ratio. This is related to changes in soil moisture and redox potential and leads to increased nutrient and pollutant retention. Small wetlands have a high perimeter length per unit area and therefore expected to have a higher water quality. However, this depends on geology, topography and wetland frequency of inundation. Furthermore, more wetland area is exposed which leads to periodic wet-dry cycles linked to nitrification-denitrification processes. Studies have demonstrates that the dynamic water retention hamper P-retention because of the increased extractable soil P (Marton et al. 2015).
- Biogeochemical activity (e.g. nitrification, denitrification, sedimentation) is also increased when hydrological connectivity is limited because of longer residence times, which thus contribute to water quality (Cohen et al. 2016; Marton et al. 2015; Powers et al. 2012).

Considering the ES-cascade of Haines-Young and Potschin (2010) often the ecosystem properties are used to describe the water purification ES, such as the total N, P or turbidity. The downside is that the contribution to the regulation of water quality is not measured. Water purification can also be measured regarding ecosystem functions through changes in nutrients (for instance differences between in- and outlets) of decomposition rates (Boerema et al. 2017; Rebelo et al. 2018). Indicators could be changes in annual average concentration, in total maximum daily load and relative and absolute changes in concentration. Changes after extreme and in the range of rainfall events are also common (Brauman et al. 2007). In the study of Jordan et al. (2003) they made use of automated flow proportional sampling to monitor nutrient removal in a restored wetland that measured the quality of the inflowing runoff and outflowing water. Rebelo et al. (2018) developed a catchment-scale mass balance sampling approach for several water quality parameters over degraded and pristine sections of palmiet wetlands. For each sampling point the quantity (mg/s) of every water quality parameter was calculated. Doherty et al. (2014) measured contaminant removal (suspended solids, P and N) by analysing water samples from stormwater above weir inverts in wetlands. And Braskerud (2002) examined nitrogen retention in constructed wetland during seven years. Samples were taken regularly on the in- and outlet of the wetlands and were analysed for N, NO_3 and NH_4.

Quantification of water purification of wetlands is challenging and costly, because of the spatial and daily, seasonally and interannual variations of rainfall as variations amongst as well as within wetlands themselves (Brauman et al. 2007; Jordan et al. 2003; Rebelo et al. 2018). One needs to take consideration between full water and nutrient budgets and deficient rapid assessments of water quality (Rebelo et al. 2018).

Models also allow to assess nutrient budgets to estimate the effect of water purification (Brauman et al. 2007). Liquete et al. (2011) estimated the effect of rivers, streams and lakes on water purification by using a nutrient transport model (Grizzetti and Bouraoui 2006). Two indicators were derived that describe (1) the capacity of nitrogen removal which is the portion of N that is naturally retained by rivers and (2) the flow of N is derived from the product of the first indicator and the total N loading in the river. The degree of nutrient removal depends on residence time, discharge, width and volume of the water body.

In addition to water provision, InVEST also contains a model for water purification and is applicable on a higher spatial resolution. This has been applied by for instance Bai et al. (2011) and Terrado et al. (2014) in a watershed. Water purification was assessed for N and P-retention. For each cell amount of exported nutrients from that cell was estimated by the annual average nutrient fluxes and the retained amount of nutrients was derived from retention coefficients which are a function of the land use/land cover (LULC). Additional input data include topography, LULC, soil depth and water yield. The output consists of the nutrient export and retention (both in kg/y). InVEST also provides the possibility to add a benefit term, namely the nutrient loads exceeding the drinking water standard and nutrient loads exceeding ecological standards. This way water purification is representative for water quality. The output gives than an estimation for nutrient retention for water quality. InVEST is capable of model other emissions, but this data on loading and filtration rates is often not available.

Water purification has also been valued economically as by Trepel (2010) who assessed the cost-effectiveness of water purification (N retention) in wetlands in Germany. Cost-effectiveness equals the cost of a load reduction of one kg N in a certain time period. The WETTRANS model was used to assess nitrogen flows across the wetland. Input data includes of information on the wetland dimensions, precipitation, evapotranspiration, substrate type, hydrological conditions in the lateral drainage basin adjacent to the wetland, water management and land use types. The outcome of the model consists of valuation of nitrogen input and output, nitrogen retention in the wetland and its efficiency to retain nitrogen.

Lastly, benefits of restoration measures can be analysed for instance by means of the avoided cost method. Boerema et al. (2014) analysed the benefit and costs of aquatic vegetation removal as a consequence of increased runoff and peak flows. Macrophytes take up nutrients from the water (benefit) but concurrently the denitrification capacity decreases as a result of the reduced residence time (cost). The evaluation method is the net change in N and P removal and is calculated for a wet summer and a dry summer scenario. The direct effect is the amount of nutrients in the removed vegetation and the indirect effect is estimated by the denitrification efficiency which is calculated by the formula of Seitzinger which depends on the water residence time. The estimations are calculated with data from international literature. The avoid cost method is then used by means of the shadow prices for N and P-removal (€/kg). The latter is the marginal cost of technical measures necessary to obtain the water quality standards.

Current representation within ECOPLAN-SE

The ECOPLAN-SE tool contains two modules for water purification.

Nitrate leaching

Two N sources are considered. Based on the maximum permitted fertilization on light and heavy soils for every crop the expected N residue is estimated. Soil leaching sensitivity is used to estimate the leaching from atmospheric deposition.

Denitrification

Nitrogen removal by dentification is estimated by the combination of the following factors:

- Groundwater supply based on topographic indices. A correction factor is applied for the soil permeability which is based on the granule size of the soil texture.
- Denitrification rate based on residence time which is a function of soil texture;
- Denitrification rate based on MHG and MLG.

Nitrate concentration which is estimated by nitrate leaching and the effective denitrification.

2.4. Water provisioning

Current representation within ECOPLAN-SE

The ES of water provision uses the ecosystem function of infiltration and is based on the principle of demand and supply of water. The module contains two aspects: quality and quantity. Based on licensed phreatic groundwater abstractions in a radius of 12 km and the actual infiltration the relative abstraction rate is estimated. The potential pollution risk is determined based on land cover and land use. This is combined with the actual infiltration and the relative abstraction rate in order to estimate respectively the potential volume of contaminated water infiltrating in the soil and the volume of potential contaminated abstracted water.

2.5. Nutrient storage in soils

Literature

Nutrient storage is important for plant growth and soil fertility. A fertile soil "provides essential nutrients for crop plant growth, supports a diverse and active biotic community, exhibits a typical soil structure, and allows for an undisturbed decomposition" as defined by Mäder et al. (2002). The most common indicators for soil quality are total organic matter or carbon, pH, available P, bulk density, texture, available K and total N (Bünemann et al. 2018; Van Eekeren et al. 2010). Other common indicators for soil quality include decomposition rate, biological respiration and soil formation (Boerema et al. 2017). This part focuses on the nutrient storage of C, N, P and K (nutrient cycling).

Heathland degradation was already mentioned as one of the human disturbances in the last century. Substitution by coniferous plantations and agriculture diminishes groundwater recharge and increases N deposition (Aerts et al. 1995; Frouz et al. 2009; Hawley et al. 2008). Examples of heathland restoration techniques are surface vegetation management and removal, soil acidity and nutrient amelioration techniques and soil disturbance and soil removal techniques (Allison and Ausden 2004; Hawley et al. 2008). The aim of these techniques is to reduce soil organic matter, nutrient availability and pH. Common wetland restoration indicators are soil organic matter, microbial biomass, pH and total concentration of C, N and P. This is usually done by taking field samples (e.g. Aerts et al. 1995; Allison and Ausden 2004; Frouz et al. 2009; Heitkamp et al. 2008; Kooijman et al. 2016; Power et al. 2006). However, they found out that restoration of former heathlands did not reduce pH and removing P is also difficult, even with completely topsoil removal. The latter is important in reducing nutrient availability drastically (N and C), as restoration of heathlands is a relatively slow process and requires many years.

Heathland and marginal arable land were gradually being converted to pine forest since the 19th century. Scots and Corsican pines are more resistant to the pore sandy soils in the Campine region in northern Belgium and are good interceptors of N deposition, produced by the agricultural sector and from combustion of fossil fuels by traffic, industry and households, and which leads to acidification and eutrophication of the already poor soil. This has an adverse impact on soil biota, litter decomposition, nutrient leaching and release of toxic substances. Hence the conversion of coniferous forest to mixed forest in this region could be one of the main solutions for this problem. As a consequence, deposition and leaching to the groundwater is reduced. In addition, nutrient cycling was improved through better litter quality of broadleaved species. The fixation of nutrients, such as N, P, K, Ca and MG in the above biomass and the nutrient uptake by roots were higher for birch compared to pine (De Schrijver et al. 2009; Gielis et al. 2008).

Several physical, chemical and biological processes are involved in the nutrient cycling of wetlands. Physical processes encompass sedimentation and adsorption. P is an element that is attached to other particles and is deposited in wetlands. Also, the dissolved form (PO_4) accumulates in sediments by sorption and precipitation. Chemical processes include transformation processes which are dominant for N removal in wetlands (e.g. ammonification, nitrification and denitrification, see above) by microbiota. Lastly, biological processes comprise of the uptake by biota and transformation processes by bacteria. For instance, N as well as P are incorporated in biota and the majority is released after the decomposition of organic matter. These processes are influenced by presence of oxygen, climate, nutrient loading rate, water inflow rate and water retention capacity (Kostel n.d.). Nutrient storage was measured in restored wetlands by Jessop et al. (2015) by the following indicators and by means of field measurements: denitrification potential (N content), herbaceous biomass, soil organic content and organic matter decomposition. Furthermore Van Eekeren et al. (2010) measured nutrient supply through analysis of soil organic matter, measuring potential mineralization of C and N and mineralizable C and N in a field study. They found out that the latter are determined by total soil N content.

Syswerda and Robertson (2014) examined trade-offs between soil ecosystem functions for tillage and no tillage systems. Soil carbon and C/N ratios were analysed from soil samples and nitrate leaching was investigated in combination with the Systems Approach for Land Use Sustainability (SALUS) model that modelled downward water drainage. SALUS models plant growth based on several plant, environmental and management parameters and also encompasses a water balance model. Measured nitrate concentrations and the modelled water drainage were combined to estimate nitrate loss over a period of eleven years. No till systems were characterized by a lower C soil content and reduced nitrate leaching compared to conventional tillage.

Remediation of soil compaction is important as not only physical properties are affected but also microbial and bioturbation activity can be impaired. Soil compaction would lead to a reduction in N mineralization and nutrient uptake and an increase in denitrification causing N loss (Batey 2009; Batey and Killham 1986; Batey and McKenzie 2006; Breland and Hansen 1996; Gregory et al. 2015).

Also models can be used for the quantification of nutrient retention. Tomscha et al. (2019) used the Land Use Capability Indicator model (LUCI) to map N and P retention in wetlands. Unlike InVEST and ARIES, LUCI is able to assess ES on a high resolution which enables analysing local spatial patterns. They used a spatial resolution of 5 m. Input data consists of land cover, soil, topography, precipitation and evapotranspiration data. Based on this data the export coefficient approach allocates a load to each cell and analyses how water, sediment and nutrient flows move through the area. Interception of N and P can be estimated. However, it is important to mention that in this study losses and their locations are modelled instead of total quantities of ES.

Current representation within ECOPLAN-SE

This ES is related to the C stock in the soil, as more N and P are bonded to the organic matter when the C stock is higher. Also, vegetation plays a role in the nutrient content in soils as leaves reduce the C/N-ratio and woody material lead to a higher C/N-ratio. Lastly N-fixating vegetation lowers the C/N-ratio. The ECOPLAN-SE tool contains a module for nutrient soil storage which is linked to the module of carbon storage in soils. C/N and N/P ratios for each land cover class were taken over from literature and used with the carbon storage module to calculate C and P storage in soils (Vrebos et al. 2017).

2.6. Climate regulation

Literature overview

The several possibilities to measure the ES climate regulation proves the complexity of the ES. As ecosystem properties the carbon stock or standing biomass are often used as an indicator for climate regulation. Besides stocks, also ecosystem functions are measured of which carbon sequestration is one of the most popular indicators of the ES of climate regulation. In addition, GHG emissions/sequestration and net primary productivity are used as indicators. Economic valuation of climate regulation is done by means of the carbon price or social damage costs related to GHG emissions (Boerema et al. 2017). The following part will list examples of studies where these quantification measures are used.

Some studies use simple indicators in the evaluation of climate regulation. For instance Radford and James (2013) conducted a GIS analysis in order to obtain a value for carbon sequestration. They digitised tree cover from aerial photographs after which the tonnes of carbon sequestration per acre and per years was derived from the percentage of tree cover multiplied by a constant factor. In combination with a survey based assessment tool, a score was assigned to the ES in order to allow comparison between different assessed ES. Likewise Lorencová et al. (2013) made use of a combination of literature and national data for the estimation of climate regulation, whereby the carbon sequestration rate is multiplied by the area of a certain land use type. They do not take into account that carbon sequestration is influenced by many factors, such as soil conditions, vegetation type and management practices.

Le Clec'h et al. (2016) not only measured water infiltration (see above) but also used carbon stocks as indicator for climate regulation. C content and bulk density was measured in soil samples to calculate the C-stock. Also Moore and Hunt (2012) used C content to estimate carbon sequestration, in particular for constructed stormwater ponds and wetlands. Total C was analysed from soil samples and rate of carbon accumulation was predicted based on the age of the constructed systems. Howe et al. (2009) examined carbon sequestration rates in natural and disturbed wetlands by measuring total C, organic C, inorganic C, C density and bulk density from soil substrate samples taken on several moments together with measurements of soil surface elevation. They calculated the rate of carbon sequestration based on the vertical accretion.

GHG emissions were also used as an indicator for climate regulation. Hefting et al. (2006) and Hefting et al. (2013) focussed on measuring N_2O emissions and denitrification respectively in riparian buffer zones and in riparian wetlands in agricultural landscapes by means of flux chambers and incubation experiments. Smukler et al. (2010) measured C stocks as well as CO_2 and N_2O emissions from organic farmlands by means of field measurements. Emery and Fulweiler (2014) analysed fluxes of CO_2, CH_4 and N_2O emissions of two plant species in marshes by assessing the change on concentrations over time.

Ghaley et al. (2014) combined biophysical and economic assessment of climate regulation. Carbon sequestration was analysed from field samples of different production systems measuring aboveground and belowground biomass accumulation estimates supplemented with literature data. Economic valuation of carbon sequestration was estimated based on the carbon market price of the EU Trading Scheme. Wüstemann et al. (2014) investigated the additional benefits of biodiversity measures implemented in the context of the biodiversity strategy plan in Germany. Their assessment for climate mitigation in avoiding GHG emissions was based on the damage cost. They only included land use changes on peatlands and forests due to data availability reasons. Estimates for the price per ton CO_2 equivalents were derived from cost-benefit analyses of previous public projects and from literature. Also Cui et al. (2012) performed an economic valuation of wetland benefits. The average amount of carbon sequestration is known from literature and so based on the carbon tax rate a value is assigned to the absorbed CO_2 per ha.

Like the other ES, also models are used to quantify the climate regulation ES. Schröter et al. (2014) developed a spatial model to estimate the provision of this ES, thereby making distinction between the capacity of ES delivery and actual flow. Net ecosystem productivity is used as indicator for carbon sequestration and was calculated as the difference between net primary productivity derived from satellite images and soil respiration. The latter is derived from a formula where soil respiration is a function of air temperature and precipitation. Because ecosystems cannot sequester as much as there are carbon emissions, carbon sequestration capacity is accounted as a flow. The result is strongly dependent on the input data.

InVEST contains a model for carbon storage and sequestration which is used in different studies to quantify carbon sequestration in for example different land use scenarios (e.g. Pechanec et al. 2018; von Essen et al. 2019). This model aggregates carbon storage for four carbon pools (aboveground biomass, belowground biomass, soil and dead organic matter) which is derived from land use and land cover maps. This way the net amount of C per parcel can be estimated (Sharp et al. 2018). Limitations of this approach include the assumption that carbon storage is constant over time for each land use type and the accuracy and reliability of the results are dependent of the land use and land cover input data (Pechanec et al. 2018; Sharp et al. 2018; von Essen et al. 2019).

Peh et al. (2014) made use of the Toolkit for Ecosystem Service Site-based Assessment (TESSA) to estimate the net benefit of ES delivered by conversion of arable land into wetlands. TESSA guides users in the ES assessment on how to identify ES, which method can be used, which data are needed, and which steps should be taken to perform that method. It can be used for qualitative as well as quantitative assessment in biophysical and monetary units. The output consists of the comparison between two states at the particular site. For the climate change mitigation tool GHG fluxes of CO_2, CH_4 and N_2O were derived from literature and were converted to tonnes CO_2 equivalents. Then a monetary was assigned to the summation by using the carbon price. As opposed to InVEST, the TESSA toolkit does not require technical knowledge of the modelling approaches and it facilitates the use of high resolution, site-scaled data. TESSA emphasis the use of local data and engagement with stakeholders at the particular site by means of assessment and interpretation processes. The downside of the toolkit consists of the few ES that could be analysed with the valuation approaches of TESSA, complexities are not specified and it does not produce spatial outputs (Neugarten et al. 2018; Peh et al. 2014).

Current representation within ECOPLAN-SE

Carbon storage in biomass

Plants take up carbon to build up their biomass. The assumption is made that mainly forests are important for C uptake on the long term. Other vegetation types also take up C, but this is more temporary as C is restored in the soil after decomposition. The suitability of the soil for several tree species is based on soil texture, soil drainage and profile. Various factors of the tree species are used to calculate the average annual carbon sequestration: mean annual accretion without thinning, density and biomass expansion factors (BEF). As a final step we translate the total growth of biomass (m^3/ha*year) to carbon sequestration values. Therefore, we use the species specific carbon density (kg /m^3). At the end the annual long-term carbon storage in biomass is estimated through combination of the soil suitability, soil cover and the carbon storage for different uptimes (short uptimes have a higher accretion per year). This model only takes long-term carbon storage of the tree trunk without side branches and the roots into account (Vrebos et al. 2017).

$$\begin{aligned} &Carbon\ storage\ in\ biomass\ \left[\frac{ton\ C}{ha * year}\right] \\ &\qquad = \left(BEF - (BEF - 1 - subsoil\ BEF)\right) * I_{mv}\left[\frac{m^3}{ha * year}\right] * densityfactor \\ &\qquad * carbon\ conversion\ factor \end{aligned}$$

Carbon storage in soils

Soils contain more than three times the amount of C then vegetation and the atmosphere together are therefore of great importance to contribute to climate change. A distinction can be made between natural soils (e.g. forests, permanent grasslands) and soils with intensive land use where the C content is much lower. Also, management practices such as drainage and soil tillage have a negative impact.

The module for the calculation of carbon storage in the soil is based on the work of Ottoy et al. (2015) and allows to assess soil organic carbon content of a land unit (LU), which is the coupling of detailed soil profile data and landcover types. A multilevel generalisation approach was executed to match statistical data related to observed profiles and associated horizons to LUs covering 98.7% of the Flemish Region. This way an estimation of SOC can be generated for particular LUs.

The study resulted in four regression equations for the calculation of carbon soil storage up to one meter depth for several land cover types and soil hydrology (drainage and groundwater abstractions):

$$Arable\ land\ \left[\frac{ton\ C}{ha * year}\right] = \left(4.4118 + 0.2293 * \%Clay + 5.1805 * Fertilizer - 0.0047 * \frac{MLG}{100} + 3.3852 * Podzol + 6.1161 * Anthrosol + 0.0001 * Clay * \frac{MHG}{100} - 0.2460 * Clay * Fertilizer + 0.2027 * Peat\right) * 10$$

$$Grassland\ \left[\frac{ton\ C}{ha * year}\right] = \left(8.6475 + 0.0290 * \%Sand - 0.0041 * \frac{MLG}{100} + 2.2362 * Fertilizer + 0.9863 * Podzol + 4.1541 * Anthrosol + 7.3375 * Peat - 0.00004 * \frac{MLG}{100} * \%Sand\right) * 10$$

$$Forest\ \left[\frac{ton\ C}{ha * year}\right] = (15.0835 + 0.8 * \%Clei - 0.017 * \frac{MHG}{100} + 0.2341 * Slope - 6.0478 * Fagus + 3.372 * Populus - 1.1636 * Quercus + 1.9505 * Betula + 8.3097 * Anthrosol + 40.2115 * Peat + 1.7264 * Podzol - 2.8944 * Ferraris + 0.0007 * \%Clei * \frac{MHG}{100}) * 10$$

$$Different\ nature\ types\ \left[\frac{ton\ C}{ha * year}\right] = \left(13.8572 + 0.2006 * \%Clay - 0.0126 * \frac{MLG}{100} + 13.4339 * Peat + 4.2009 * Podzol - 3.5461 * Heath + 1.9306 * Pioneers\ vegetation + 2.1491 * Reedland\right) * 10$$

Ambitions for PROWATER

The ECOPLAN methods for SOC are based on local empirical data and innovative statistical techniques that are hard to improve (Ottoy et al. 2015).

But evidently there are other GHG-emissions (methane, nitrous oxide) that are not accounted for changes in SOC-stocks following land-use change. Modelling these is very challenging and considered to be outside the scope of PROWATER.

The same accounts for GHG emissions from wetlands and surface waters. The temporal or permanent retention of water in wetlands is beneficial for denitrification (water purification), water provision and carbon sequestration, but may have negative consequences for climate regulation. Denitrification occurs in anoxic conditions (water saturated zone) and may emit considerable amounts of N_2O in the atmosphere. Wetlands are also a natural source of CH_4. Lowering the summer water level on the other hand will decrease emissions of these GHG but will enhance CO_2 emissions (Blackwell and Pilgrim 2011; Clarkson et al. 2014; Kayranli et al. 2010). However, Wüstemann et al. (2014) demonstrated that this trade-off for CH_4 and CO_2 is stronger for the decrease in CO_2 after rewetting than the increase of CH_4. Again, this is challenging and requires dedicated research. We may take into account hydroperiod indicators for wetlands to assess whether the risk for methane emission is high or low. Grasping hydrological behaviour of the landscape is the first step towards a credible assessment of climate mitigation dynamics

3. Synthesis and conclusions

Human pressures, such as land use changes, soil sealing, groundwater abstractions and drainage have had an enormous impact on the hydrological system, leading to increased peak flows, declining groundwater levels and a decreased natural water availability. In combination with climate change, threatening water security will become a key. Also, other ecosystem services, such as soil nutrient retention, soil carbon sequestration and biodiversity are affected. When strategic water reservoirs and/or aquifers are sufficiently replenished, drought periods and associated water demands can be bridged. However, the replenishment of these strategic water reserves has become insufficient because our landscapes have been degraded and are not adapted to deal with extreme weather.

The aim of PROWATER is to build resilience against droughts, water scarcity and extreme precipitation events. The objective is to implement ecosystem-based adaptation measures that increase the retention and infiltration capacity of the landscape by restoring ecosystems and enhancing natural processes. With PROWATER we focus on specific types of measures that improve soil permeability through agricultural soil management, reduce interception through forest conversion/management practices, promote and prolong water storage in floodplain wetlands, promote deferred infiltration through restoration of upstream depressional wetlands and remediate soil sealing impacts through infiltration ponds.

It is crucial to assess the impact of these EbA measures on the ES related to PROWATER in order to develop deliberate and correct guidelines for the future implementation of these measures. Besides their key impact on water regulation and provision, these measures may deliver many additional benefits. For many ecosystem services, their supply is driven by (complex interactions of) ecohydrological processes.

This report starts with an overview of the ES and EbA measures, followed by a comprehensive literature review of ES quantification methods where the main principles, methods and their feasibility are discussed. Emphasis is placed on water regulation that is approached as an encompassing concept, discussing the different water budget components. A selection of the ES quantification methods is used in the ECOPLAN-SE tool to model the impact of EbA measures on the provision of different ES. Besides an overview of the current models in ECOPLAN, modelling approaches are discussed that need to be developed, whereby ECOPLAN-SE can be extended to improve and deliver more detailed and accurate assessments of the impact of EbA measures.

The effectiveness of a measure therefore always depends on its context. For example, conversion from coniferous to deciduous woodland on naturally wet soils will obviously not result in infiltration. Also, wetland creation will not result in groundwater recharge if the subsoil has a too low permeability and improving soil quality of the topsoil will have limited effect if subsoil is compacted. The table thus provides general insights into the effects that the measures can have on the various services. However, it is recommended to read the relevant sections of the report to really find out which factors interplay to the effectiveness of measures. The final tool will take these nuances into account. This is the exact reason why we need modelling approaches that take into account the biophysical processes that drive ES-supply.

Table 11 gives a concise overview of the general impact of the EbA practises on the different ES. However, it must be stressed that the impact of each measure depends a lot on the local context (soil type, geology, environment, topography). The effectiveness of a measure therefore always depends on its context. For example, conversion from coniferous to deciduous woodland on naturally wet soils will obviously not result in infiltration. Also, wetland creation will not result in groundwater recharge if the subsoil has a too low permeability and improving soil quality of the topsoil will have limited effect if subsoil is compacted. The table thus provides general insights into the effects that the measures can have on the various services. However, it is recommended to read the relevant sections of the report to really find out which factors interplay to the effectiveness of measures. The final tool will take these nuances into account. This is the exact reason why we need modelling approaches that take into account the biophysical processes that drive ES-supply.

Table 11: Summarising table of the effect of the different EbA measures on ecosystem services. The impact on ES the local context (e.g. soil type, slope, soil management, and so on). This table provides a general view on the EbA measures on ES.

Ecosystem services / EbA measures PROWATER	Water regulation: Water retention	Water regulation: Water infiltration	Water purification (nitrogen removal)	Nutrient storage in soils	Erosion prevention	Climate regulation: Carbon	Climate regulation: Methane	Climate regulation: Nitrous oxide
Conversion from coniferous to broadleaved forest	Medium positive (relevant)	[1] Medium positive (relevant)	Low positive (less relevant)	Medium positive (less relevant)	Medium positive (less relevant)	[2] Low negative (less relevant)	[3] Low negative (less relevant)	[3] Low negative (less relevant)
Conversion from forest to heathland/grassland	[1] High positive (relevant)	[4] High positive (relevant)	Neutral (less relevant)	Medium negative (less relevant)	Medium negative (relevant)	[5] Medium negative (relevant)	Neutral (less relevant)	Neutral (less relevant)
Improving soil permeability (conservation tillage/cash crops) [6]	High positive (relevant)	High positive (relevant)	Low positive (relevant)	Low positive (less relevant)	High positive (relevant)	Medium positive (relevant)	Low positive (less relevant)	Neutral (less relevant)
Restoration permanent wetlands/re-meandering [7]	High positive (relevant)	Low positive (less relevant)	Medium positive (relevant)	Medium positive (less relevant)	Medium positive (less relevant)	Medium positive (less relevant)	Medium negative (less relevant)	Medium negative (less relevant)
Restoration of temporary wetlands	High positive (relevant)	High positive (relevant)	Medium positive (relevant)	Low positive (relevant)	Medium positive (less relevant)	Medium positive (less relevant)	Medium negative (less relevant)	Medium negative (less relevant)
Runoff collection through infiltration ponds or weirs in ditches	Medium positive (relevant)	Medium positive (relevant)	Medium positive (relevant)	Low positive (relevant)	Medium positive (less relevant)	Low positive (less relevant)	Low negative (less relevant)	Low negative (less relevant)

Impact		Relevance for PROWATER
Low positive	Neutral	Relevant
Medium positive	Low negative	Less relevant
High positive	Medium negative	

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[1] The effect depends on the type of soil. Forest cover and interception have a positive effect on heavy soils, as they buffer extreme precipitation events and promotes infiltration. In contrast, sandy soils in general reduce infiltration.

[2] Conifer trees have dense canopies that intercepts a certain portion of light which causes a lowering of soil temperature. This reduces in turn slow down decomposition which leads to an accumulation of organic matter and increased carbon sequestration (Barsoum and Henderson 2016).

[3] Processes responsible for methane and nitrous oxide emissions vary in time and space and depend on soil texture, topography, precipitation and nitrogen limitations (Díaz-Pinés et al. 2018).

[4] The effect is strongly dependent on the soil type, scale of planting, forest design and replaced landcover. The effect mentioned in the table is the case for sandy soils, but is different for chalk soils. Conversion from broadleaved woodland to grass has a little impact as the uptake of root water can be maintained, even during drought periods (Calder et al. 2002; Nisbet 2005).

[5] The C stock under grassland can be at the same level as under forest, provided that the grassland is permanent and natural grassland or either extensive grassland with livestock.

[6] Deep compaction is not taken into account in PROWATER but must be investigated as it can be a problem.

[7] Restoration of permanent wetlands and rivers involves several measures and mostly include riverbank stabilisation, which has a positive effect on water quality, nutrient storage in soils, erosion control and carbon sequestration.

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