Science of the Total Environment 639 (2018) 1610–1622

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Topsoil structure stability in a restored floodplain: Impacts of fluctuating water levels, soil parameters and ecosystem engineers

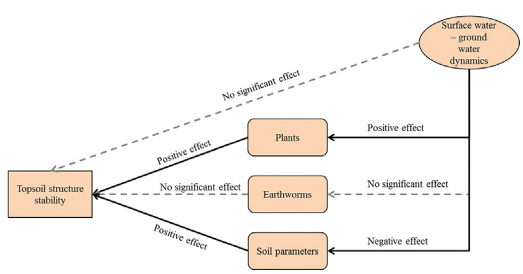
A. Schomburg [a,b,*], O.S. Schilling [b,c], C. Guenat [d,e], M. Schirmer [b,f], R.C. Le Bayon [a,1], P. Brunner [b,1]

[a] Functional Ecology Laboratory, Institute of Biology, University of Neuchâtel, Switzerland
[b] Center for Hydrogeology and Geothermics (CHYN), University of Neuchâtel, Switzerland
[c] Department of Geology and Geological Engineering, Université Laval, Pavillon Adrien-Pouliot, Québec, QC G1V 0A6, Canada
[d] Laboratory of Ecological Systems – ECOS-EPFL, Ecole Polytechnique Fédérale de Lausanne (EPFL), Switzerland
[e] WSL Swiss Federal Institute for Forest, Snow and Landscape Research, Site Lausanne, Switzerland
[f] Eawag, Swiss Federal Institute of Aquatic Science and Technology, Water Resources & Drinking Water, Dübendorf, Switzerland

HIGHLIGHTS

- Numerical flow modelling is used to analyse water fluctuations in a floodplain soil.
- Full and partial water saturation can affect macro-biological soil engineering.
- Fluctuating water levels can indirectly affect topsoil structure stability.
- Fluctuating water levels can positively affect root size and root abundance.
- Topsoil structure stability can be improved under more favourable soil parameters.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 19 March 2018
Received in revised form 9 May 2018
Accepted 9 May 2018
Available online xxxx

Editor: D. Barcelo

Keywords:
Soil aggregate
Structural equation modelling
Earthworm
Pioneer Plant
Flood
Surface water-groundwater modelling

ABSTRACT

Ecosystem services provided by floodplains are strongly controlled by the structural stability of soils. The development of a stable structure in floodplain soils is affected by a complex and poorly understood interplay of hydrological, physico-chemical and biological processes. This paper aims at analysing relations between fluctuating groundwater levels, soil physico-chemical and biological parameters on soil structure stability in a restored floodplain. Water level fluctuations in the soil are modelled using a numerical surface-water–groundwater flow model and correlated to soil physico-chemical parameters and abundances of plants and earthworms. Causal relations and multiple interactions between the investigated parameters are tested through structural equation modelling (SEM). Fluctuating water levels in the soil did not directly affect the topsoil structure stability, but indirectly through affecting plant roots and soil parameters that in turn determine topsoil structure stability. These relations remain significant for mean annual days of complete and partial (>25%) water saturation. Ecosystem functioning of a restored floodplain might already be affected by the fluctuation of groundwater levels alone, and not only through complete flooding by surface water during a flood period. Surprisingly, abundances of earthworms did not show any relation to other variables in the SEM. These findings emphasise that earthworms have efficiently adapted to periodic stress and harsh environmental conditions. Variability of the topsoil structure stability is thus stronger driven by the influence of fluctuating water levels on plants than by the abundance of earthworms. This knowledge about the functional network of soil engineering organisms, soil parameters and fluctuating water levels and how they affect soil structural stability is of fundamental importance to define management strategies of near-natural or restored floodplains in the future.

* Corresponding author at: Functional Ecology Laboratory, Institute of Biology, University of Neuchâtel, Switzerland.
E-mail address: andreas.schomburg@unine.ch (A. Schomburg).
[1] Co-last authorship.

https://doi.org/10.1016/j.scitotenv.2018.05.120
0048-9697/

1. Introduction

The development and preservation of a stable soil structure are fundamental aspects to ensure the functioning of an ecosystem (Bronik and Lal, 2005). The soil structure of young and dynamic or recently restored ecosystems, such as floodplains, is highly susceptible to external interferences. Regular flooding, erosion and sediment deposition can result in the decrease of soil structural stability or reset initial soil stabilisation processes (Junk and Welcomme, 1990; Bätz et al., 2015; Graf-Rosenfellner et al., 2016). However, the conservation of biodiversity in floodplains is directly linked to the structural stability of the floodplain soil, developing a mosaic of various floodplain habitats (Malmqvist and Rundle, 2002). Ecosystem services such as nutrient and pollutant recycling, flood and erosion control, carbon storage as well as surface and groundwater purification are promoted by stable conditions in floodplains (Brooks and Shields, 1996; Diaz-Zorita et al., 2002; Acreman et al., 2003; Bronik and Lal, 2005). Soil structure stability has been shown to be improved mainly by soil organisms' activities. In floodplain ecosystems, physical processes such as the alternation of desiccation and remoistening of the soil matrix are an additional factor driving its structural stability (Angers and Caron, 1998; Bronik and Lal, 2005). Furthermore, soil structure stability is directly influenced by various soil physico-chemical parameters, such as the soil texture (Tisdall and Oades, 1982; Sollins et al., 1996; Bullinger-Weber et al., 2007), carbonate and organic matter contents (Tisdall and Oades, 1982; Chorom et al., 1994; Haynes and Naidu, 1998; Kong et al., 2005). To a large extent, soil physico-chemical parameters can also influence soil organisms structuring the soil, especially through pH values (Ivask et al., 2007) and soil texture (Bullinger-Weber et al., 2007; Fournier et al., 2012). On the macro-biological scale, plants and earthworms are among the most successful soil ecosystem engineers in temperate ecosystems, being able to efficiently improve soil structure stability through the formation of water-stable macro-aggregates (Brown et al., 2000; Kong and Six, 2010). Plants physically enmesh soil particles or provide inputs of organic matter through root exudates that cement particles and/or stimulate the production of bonding agents by soil microbes (Degens et al., 1994; Angers and Caron, 1998). Earthworms build up soil aggregates through burrowing and casting as they move into the soil, and agglutinate organic and mineral particles with mucus and saliva produced in their digestive system (Blanchart et al., 1997; Brown et al., 2000; Lavelle and Spain, 2001). Both plants and earthworms are considered as particularly efficient in floodplains due to their wide-ranging strategies to tolerate harsh environmental conditions (Gurnell and Petts, 2002, 2006; Thonon and Klok, 2007; Crouzy and Perona, 2012; Perona et al., 2012). Nevertheless, plant abundance and earthworm populations generally tend to be reduced in habitats frequently threatened by floods (Plum and Filser, 2005). So far, the effect of flood frequency on plants and earthworms has been described indirectly through sampling at varying distances to the river (Salomé et al., 2011; Bullinger-Weber et al., 2012; Fournier et al., 2015), or by monitoring and counting the number of flood events impacting the studied habitats (Ausden et al., 2001; Zorn et al., 2005; Gurnell and Petts, 2006; Ivask et al., 2007; Corenblit et al., 2009). With the methods applied so far, information concerning the response of ecosystem engineers towards floods is restricted to conditions in which soils are completely submerged. However, soil moisture contents below full saturation have also been shown to affect ecosystem engineers, as presented by Wever et al. (2001) and Davis et al. (2006) for earthworms and for plants (Henszey et al., 2004). Given that the degree of saturation in the soil is heavily influenced by the depth to groundwater, water table fluctuations could be an important control on the development of soils. A shallow depth to groundwater is likely to occur far more frequently than a flood event submerging the surface (Fig. 1). As ecosystem engineers are most widespread in the upper 20 cm of soil, representing the major part of the drilosphere and the rhizosphere (Lavelle et al., 1997; Tanner, 2001), water saturation through rising groundwater in this zone might already have a significant impact on their abundance and their capacity to improve the topsoil structure stability. To our best knowledge, there have not been any investigations analysing the relations between fluctuating groundwater levels, ecosystem engineers and soil parameters with regard to the stability of the topsoil structure. As the occurrence of ecosystem engineers and the development of the topsoil structure in a floodplain might be the result of the high temporal variability of groundwater fluctuations over seasons and years, groundwater- and soil moisture monitoring needs to be carried out over a time span of several years in order to estimate its mean impact at an intermediate time scale. For a number of reasons, numerical models simulating the spatial and temporal dynamics of surface water (SW)–groundwater (GW) interactions in the river, the underlying aquifer and the floodplain can provide crucial information in this endeavour: I) a well calibrated flow model is a physically-based interpolator that allows quantifying groundwater fluctuations in space and time. Consequently, II) profiles of water saturation can be analysed at any point within the simulated floodplain. Numerical models can therefore complement the available observation data to a degree which is not possible employing additional monitoring equipment. However, the uncertainties of the numerical model need to be considered. A modelling approach to link the different variables to soil stability is further required. Structural equation modelling (SEM) has been described as a pertinent statistical method for the identification of causal effects among variables as well as direct and indirect linear relations (Bollen, 1989). More recently, SEM were improved for the analyses of more complex causal relations and multiple interactions (Grace and Bollen, 2006, 2008; Shipley, 2016). Using SEM, causal relations of biological factors and soil structure have been demonstrated for different ecosystems by Rillig et al. (2002) and Chaudhary et al. (2009). In this study, we aim to identify the relations between fluctuating water levels, plants, earthworms, soil parameters, and their combining effects on soil structural stability through SW-GW flow modelling and SEM. We hypothesise that (I) water level fluctuations in soils can explain the degree of topsoil structure stability through the distribution patterns of ecosystem engineers and/or the soil physico-chemical parameters. We assume concomitantly that (II) partial and complete water saturation might directly impact topsoil structure stability and indirectly ecosystem engineers' abundance. We furthermore expect to identify (III) direct effects of ecosystem engineers and soil parameters on the topsoil structure stability, including interactions between them.

2. Material and methods

2.1. Study site

The study was conducted at the "Schaffäuli" floodplain site along the restored section of the Thur River close to Niederneunforn (NNF, 8°77′ 12″ E, 47°59′10″ N) in the Canton Thurgau, Switzerland (Fig. 2; Schirmer et al., 2014). The Thur River is the longest river in Switzerland without any regulation by artificial reservoirs or natural lakes over the entire water course length of approximately 130 km (Fournier et al., 2013). However, the river is canalised and embanked along most of its course. The source of the river is located in the limestone-dominated Säntis massif at 2500 m a.s.l. with a catchment area of 1750 km^2 (Samaritani et al., 2011). The hydrologic regime is nivo-pluvial. For the period 1910–1999 discharge at the Neunforn gauging station (Canton Thurgau river gauging station no: F2900, coordinates: 8°46′57.78″ E, 47°35′20.76″ N, altitude: 372 m a.s.l.) was between 3 and 1100 m^3 s^{-1} with a mean annual discharge of 46.8 m^3 s^{-1} (Horat and Scherrer AG Hydrologie und Hochwasserschutz, 1999). Flood events frequently occur during the snow melt period in spring and after intense rainfall events in autumn leading to inundations of the floodplain (Fournier et al., 2013). Maximum discharges at the Andelfingen gauging station, 10 km downstream of NNF (Swiss Federal Office for the Environment river gauging station no: 2044,

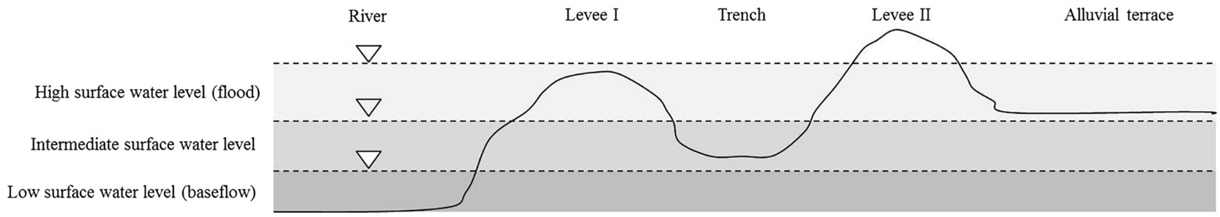


Fig. 1. Example of surface water–groundwater interactions in a floodplain. The white triangles represent the water level relative to water discharge. During intermediate surface water level, groundwater submerges the trench and saturates the topsoil of the alluvial terrace, without surface water directly inundating the trench and the alluvial terrace, as it is blocked by levee I and II. Only at high surface water level (flood), the trench is directly flooded by surface water; the alluvial terrace is inundated through rising groundwater.

coordinates: 8°40′55.08″ E, 47°35′47.46″ N, altitude: 770 m a.s.l.) were measured in 1910 (1190 m^3 s^{-1}), in 1978 (1060 m^3 s^{-1}) and in 1999 (1150 m^3 s^{-1}) (Horat and Scherrer AG Hydrologie und Hochwasserschutz, 1999). The last large flood event measured at the station Neunforn occurred in 2013 and peaked at 1017 m^3 s^{-1}, leading to sediment accumulations of up to 30 cm. As a result of the flood in 1999, the "Schaffäuli" site was restored by removing the embankments of the right river bank over a section of 1.5 km (Fig. 2). Since then, the adjacent floodplain has been equipped and systematically monitored for surface water and groundwater flow as well as for vegetation developments (e.g. Vogt et al., 2009; Schirmer et al., 2014). Since the restoration, the dynamics of both erosion and sedimentation increased strongly, and a gravel bank of temporally and spatially varying extent formed downstream of the embanked zone. Diem et al. (2014) identified four zones (K_1 to K_4) of distinct hydraulic conductivities within the "Schaffäuli" study site:

(1) The highly permeable alluvial gravel island/riverbank section that is located along the restored section ($K_1 = 6 \times 10^{-2}$ m s^{-1}),
(2) slightly less permeable sediments in the alluvial forest and riverbed surrounding the alluvial gravel island ($K_2 = 2 \times 10^{-2}$ m s^{-1}) (reduction of 66% compared to K_1),
(3) equally permeable alluvial sediments encompassing the alluvial forest along the embanked section ($K_3 = 1 \times 10^{-2}$ m s^{-1}) (reduction of 85% compared to K_1), and
(4) less permeable sediments of the embankment all along the left river bank ($K_4 = 4 \times 10^{-3}$ m s^{-1}) (reduction of 95% compared to K_1).

2.2. Field sampling design and soil sampling

Nine sampling plots were selected in the restored floodplain section with varying distances to the river (12–53 m). The surface area of each plot was adapted to the local extent of the vegetation and subsequently normalised to a square-shaped standard area of 25 m^2. The locations of sampling plots are illustrated in Fig. 2. The vegetation and soils were described and sampled in each plot in July 2015, including five replicates that were randomly taken within each plot. Soil types were described as Calcaric Fluvic Arenosols, Calcaric Fluvisols or Calcaric Fluvic Cambisols (IUSS Working Group WRB, 2015). Vegetation was recorded in relevees according to the method of Braun-Blanquet (1964). The percentage of herbs and shrubs covering the soil surface was estimated and subsequently used for the calculation of the root size and root abundance factors (see Section 2.3). During soil description, composite soil samples and macro-aggregates > 250 μm (Tisdall and Oades, 1982; Six et al., 2004) were uniformly taken from each replicate individually due to numerous soil layers originating from past flood events. Topsoil layers represented an alternation of A(k) and C(k) horizons for Fluvisols and Arenosols and of A(k) and Bw(k) for Cambisols. The thickness of the

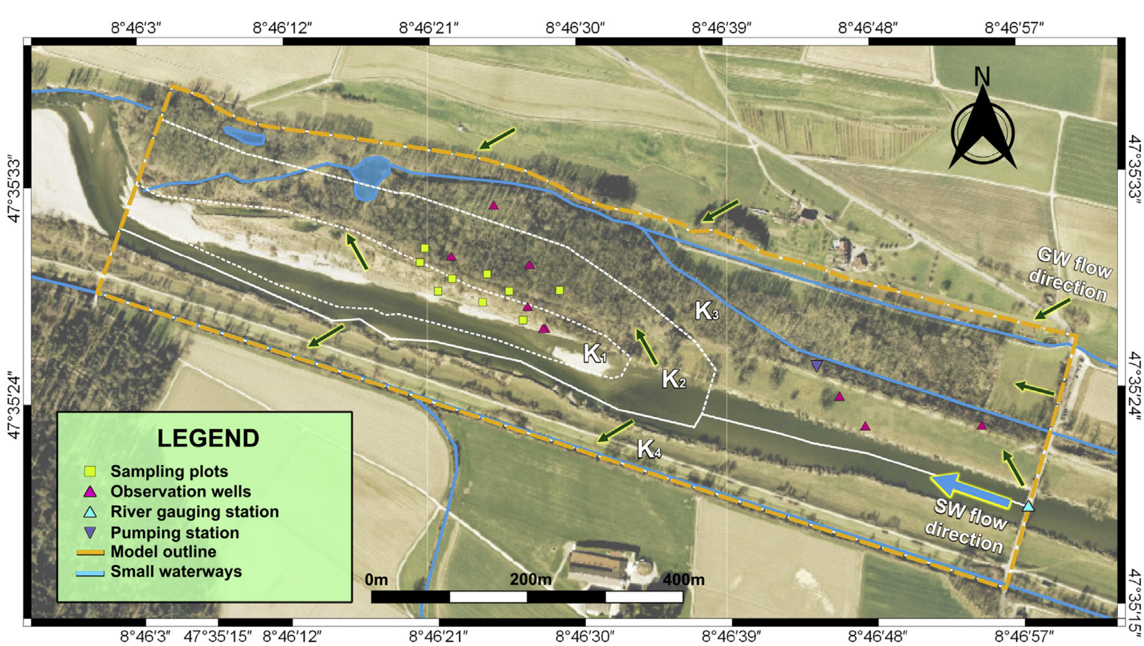


Fig. 2. Map of the different sampling locations. The flow direction of the Thur River is indicated by the big arrow within the river, GW flow directions are indicated by the smaller arrows. The embanked river section is located on the right border of the river upstream of the K2 zone, and the restored river section starts between the upstream limits of the K1 and the K2 zones. K-zones are separated by the dotted lines. The spatial extent of the numerical flow model is indicated by the dashed orange line. Coordinate system: WGS84. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)
Orthophotos reproduced with permission of swisstopo (BA17095).

A(k) horizons was limited to maximally 3 cm, so representative sampling could not be performed on horizons separately. The depth of all samples was limited to the upper 20 cm of soil representing the major part of the drilosphere and the rhizosphere. Soil samples and macro-aggregates were dried at 40 °C in an oven for 72 h, homogenised and sieved at 2 mm.

2.3. Sampling of ecosystem engineers

Plant roots and earthworms were sampled in October 2015 and described at the same sites at which soil sampling was performed. Plant root size and abundance were used as indicators in the upper 20 cm of soil and described according to the guidelines for soil description (IUSS Working Group WRB, 2006). Digits between 1 (predominantly very fine roots < 0.5 mm in diameter) and 4 (predominantly coarse roots of >5 mm in diameter) were assigned for the root size. Digits between 0 (no roots) and 4 (with >200 in the total number of roots smaller than 2 mm or with >20 in the total number of roots larger than 2 mm) were assigned for the root abundance (IUSS Working Group WRB, 2006). Obtained digits were subsequently normalised by the proportions of shrub and herbaceous plant roots that were determined during the vegetation relevee. The normalised values represent root size and root abundance factors, and were used as explanatory variables in the subsequent statistical analysis.

Earthworms were extracted in autumn 2015 using the hot mustard extraction on a 0.5 × 0.5 m square combined with the hand-sorting method on a soil cube of 0.2 × 0.2 × 0.2 m dug out in the center of the plot (Bouché and Aliaga, 1986; Lawrence and Bowers, 2002). The extracted earthworms were stored in 70% ethanol. Adult earthworms were weighed, identified to species level according to the identification key of Blakemore (2008) and classified according to their ecological categories (Bouché, 1972). Juveniles were just grouped and weighed. Total and mean earthworm abundance (individuals per m^2) was calculated for each sampling site.

2.4. Analysis of soil physico-chemical parameters

Soil texture was determined using the pipette-Robinson method. For soil chemical parameters, pH was measured using a combined pH Meter/conductometer (914 pH Meter/conductometer, Metrohm, Herisau) with the soil/water ratio of 1:2.5. Cation exchange capacity (CEC) and base saturation (Bs) were analysed using a buffered barium chloride solution (pH 8.5) that was further diluted 500 times and analysed for K, Mg, Ca and Na using an Inductively Coupled Plasma–Optical Emission spectrometer (ICP-OES Optima 2100 DV, Perkin Elmer). Total carbonate content ($CaCO_3$ tot) was measured using a Bernard Calcimeter described in Vatan (1967). The amount of active carbonates ($CaCO_3$ act) was determined according to the method of Drouineau (1942). Organic carbon and total nitrogen contents were measured using a CHN analyser (CHN-O Analyzer, Flash 2000, Thermo Scientifics).

2.5. Soil structure stability analysis

The degree of topsoil structure stability was identified by the analysis of aggregate stability which is used as the main indicator for soil structure stability (Six et al., 2000). The percentage of water-stable aggregates was determined for macro-aggregates of 250–2000 μm size according to the method described in Kemper and Rosenau (1986). A modified automatic diving apparatus equipped with sieves of 250 μm mesh size was applied for this analysis (Murer et al., 1993).

2.6. Hydrological flow model

2.6.1. Model setup

As a typical example of a surface water- groundwater system, SW and GW are characterised by dynamic exchange fluxes at the NNF site. The potential for flooding is therefore not only controlled by river flow and riverbed topography, but also by GW levels and the hydraulic conductivity (K [L/T]) of the riverbed and the underlying aquifer (Winter et al., 1998; Brunner et al., 2009; Schilling et al., 2017b). As exchange fluxes cannot be easily measured, physically-based, fully-coupled simulations of SW and GW were useful for the dynamic computation of exchange fluxes and flooding under consideration of all the relevant hydrological and physical parameters. For this purpose, the physically-based and fully-coupled SW-GW flow model HydroGeoSphere (HGS) (Therrien et al., 2010; Brunner and Simmons, 2011; Kurtz et al., 2017) that has been successfully applied to simulate complex river-aquifer systems (e.g., Banks et al., 2011; Karan et al., 2014; Schilling et al., 2014, 2017a, 2017b; Tang et al., 2015, 2017), was used to simulate the NNF site. HGS allows the coupled simulation of SW flow based on the diffusion-wave approximation of the 2-D Saint Venant equation and of variably-saturated GW flow based on the Richards equation and the van Genuchten parametrisation. Specific details on the underlying equations and parameters can be found in the HGS manual (Aquanty Inc., 2016).

The spatial extent of the simulated area is 0.37 km^2. The aquifer in NNF is approximately 6 m thick and consists of glacio-fluvial sandy gravels (Vogt et al., 2009; Vogt et al., 2010; Diem et al., 2014). The aquitard is composed of a lacustrine clay and its vertical limits were estimated by Diem et al. (2014). Following the procedure described by Käser et al. (2014), the horizontal numerical grid for the NNF site was generated using AlgoMesh (HydroAlgorithmics Pty. Ltd.) and GridBuilder (McLaren, 2011), and consisted of 8642 approximately equilateral finite elements. Within the river, the elements had an average side length of 7 m, in the alluvial forest of the restored section 15 m and in the alluvial forest of the embanked section 25 m. Vertically, the model was discretised into 10 grid layers with proportional thicknesses: 1.3% of the total vertical extent for each of the four topmost layers, 13% for each of the following four layers, and 23.8% for the bottom two layers. The fine vertical discretisation close to the surface was employed to guarantee numerical integrity in solving the highly non-linear Richards equations for unsaturated flow in the soil.

Compared to embanked river sections, both the riverbed topography and its hydraulic conductivity can be highly transient in the natural and restored river sections. This is due to the pronounced erosion and deposition processes in these sections (Gianni et al., 2016; Partington et al., 2017). The riverbed topography of such systems is thus associated with structural uncertainties which, however, can be reduced using approaches such as high-resolution through-water photogrammetry (Feurer et al., 2008; Brunner et al., 2017; Schilling et al., 2017a). At the NNF site, a digital elevation model (DEM) dating from 01 Feb. 2015 with a horizontal resolution of 0.5 m was available for the floodplain and the embanked section from the Amt für Raumentwicklung Kanton Zürich (http://www.geolion.zh.ch/geodatensatz/show?gdsid=298, accessed 03 Aug. 2017). However, for the restored river section, no high-resolution riverbed DEM was available. The topography of the riverbed for that section was therefore linearly interpolated between measured cross-sections.

2.6.2. Boundary and initial conditions

Transient simulations of four years (2013, 2014, 2015 and 2016) were carried out based on daily changing boundary conditions (Fig. 3). The Thur River was implemented as a second-type (specified flux) boundary condition using discharge measurements from the Neunforn gauging station (Fig. 3b). The small existing data gaps were bridged based on discharge data from the nearby Andelfingen gauging station. River outflow was simulated using a critical depth boundary condition. Precipitation was conceptualised as a specified nodal flux boundary condition using measurements from the NNF rain gauge (MeteoSchweiz, station: NIE, coordinates: 8°47′ E, 47°36′ N, altitude = 440 m a.s.l., Fig. 3a). Evapotranspiration was not simulated. During flood events it is not important (see Diem et al., 2014). Nevertheless,

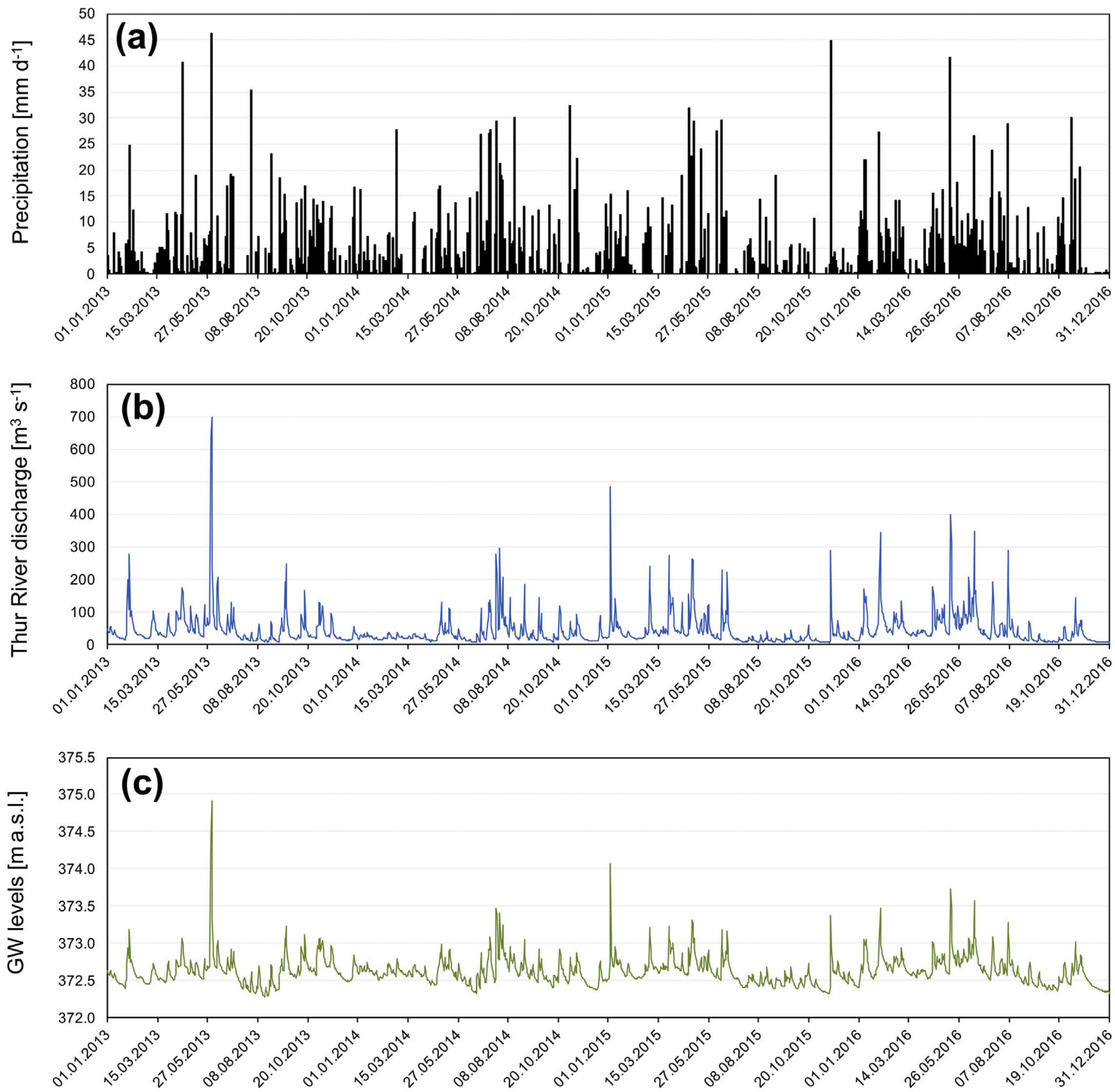


Fig. 3. Time series of (a) precipitation (daily sums, MeteoSchweiz, station: NIE), (b) River Thur discharge (daily averages, Canton Thur, station F2900) and (c) GW levels of the most upstream piezometer (daily averages, Canton Thur, station: G2905). These time series were used to define the boundary conditions of the numerical flow model.

during shallow groundwater conditions evapotranspiration could constitute a noteworthy flux. However, given the good hydraulic connection with the river, water leaving the system through evapotranspiration will be immediately replaced by infiltrating river water. Inflow of GW on the upstream model boundary (South-East) was implemented as a specified head boundary condition using hydraulic head measurements (Fig. 3c) from the most upstream piezometer (see Fig. 2). Closely following the conceptualisation of Diem et al. (2014), outflow of GW on the downstream model boundary (North-East) was implemented as a fixed-head boundary condition of 370.2 m a.s.l., and GW outflow on the lateral model boundaries (South-West and North-East) was defined as a specified flux boundary condition (nodal volumetric flux of -10 m^3 day^{-1}). The inflow of river water to the restored river section was monitored at the Neunforn river gauging station with a 5-minute interval. Groundwater levels within the floodplain were monitored along two transects and multiple additional piezometers scattered throughout the floodplain at a 15-minute interval (see Fig. 2 for the detailed locations of the different observation points). The value of the different hydraulic parameters used for the numerical flow simulations of the NNF site are listed in Table 1. The two surface flow parameters "rill storage height" and "Manning's coefficient" were calibrated to reduce the residual between the simulated and measured surface water depth at the Neunforn gauging station.

2.7. Mean annual days of water saturation (MADOWS)

For each sampling plot, vertical water saturation profiles were extracted from the numerical flow model and used as an indicator for

Table 1
Parameter values used for the numerical flow model.

Parameter	Value	Source
Saturated hydraulic conductivity (K)	$K_1 = 6 \times 10^{-2}$ m s^{-1} $K_2 = 2 \times 10^{-2}$ m s^{-1} $K_3 = 1 \times 10^{-2}$ m s^{-1} $K_4 = 4 \times 10^{-3}$ m s^{-1}	Diem et al. (2014)
Porosity (n)	0.2	Diem et al. (2014)
van Genuchten α	3.48 m^{-1}	Li et al. (2008)
van Genuchten β	1.75	Li et al. (2008)
Rill storage height	0.1 m	Calibrated
Manning's coefficient	0.0525 s m$^{-1/3}$	Calibrated

groundwater level fluctuations. The water saturation was averaged to one daily value for the upper 20 cm of the soil (i.e., the topsoil). For each year, the number of days with complete water saturation (100%) and partial water saturation (>85%, >70%, >55%, >40%, >25%) were counted. Complete water saturation can be either a result of a direct inundation by surface water or of groundwater saturation through rising surface water levels (Fig. 1). Based on these counts per year, one average value for the mean annual days of water saturation per saturation level (MADOWS100, MADOWS85, MADOWS70, MADOWS55, MADOWS40 and MADOWS25) was calculated for the period 2013–2016.

2.8. Statistical analysis

2.8.1. Variance analysis

Differences between the water saturation levels were analysed with a one-way analysis of variance (ANOVA). Requirements for ANOVA application (normal distribution and variance homogeneity) were tested in advance using the Shapiro-Wilk test and the Levene test, respectively. Group specific differences were subsequently determined using Tukey's HSD posthoc analysis. Differences were considered significant at a $P < 0.001$ level.

2.8.2. Structural equation modelling (SEM)

Full structural equation models (SEMs) were set up in two steps to estimate direct and indirect effects of variables on the degree of soil structure and to determine complex causal relations among these variables (Bollen, 1989; Grace and Bollen, 2006; Shipley, 2016). A priori, conceptual models in form of two directed acyclic graphs (DAGs, Shipley, 2016) were established under consideration of biologically plausible and relevant hypotheses and pathways (Fig. 4a, b). The direction separation (d-separation) method was then applied to verify the plausibility of the pathways drawn in the DAGs based on Fischer's C test and the corrected Akaike information criterion (AICc) (Shipley, 2016). Topsoil structure stability (labelled by "y", Fig. 4a, b) was determined as the response variable in all models with regard to the hypotheses (see Section 1). Six models were run with MADOWS as the main predictor variable separated according to complete (MADOWS100) and partial water saturation (MADOWS85, MADOWS70, MADOWS55, MADOWS40, MADOWS25) (labelled by x_1). Observations for soil physico-chemical properties and ecosystem engineers were assembled as composite variables (Grace and Bollen, 2006, 2008). Composite variables are endogenous variables that are statistically similar to a predictor variable in multiple regressions. They represent a weighed combination of their associated variables and neglect variables that are not statistically significant. Soil physico-chemical properties were assembled as the composite variable "soil parameters" (labelled by η_3), root size and root abundance factors for herbaceous and shrub roots were assembled as the composite variable "plants" (labelled by η_1), and abundances of earthworms of different ecological categories were assembled as the composite variable "earthworms" (labelled by η_2). For soil texture, which was assembled into "soil parameters", clay content was neglected in order to avoid statistical dependencies among predictor variables. Path analyses of the relations suggested in the conceptual models (i.e., in the DAGs) were performed using SEM with maximum likelihood estimation for Chi-squared analysis (Shipley, 2016). Path coefficients were calculated for each predicted relation in order to define each relation's individual strength. Path coefficients are labelled with the letters β, γ, ι, κ and λ and grouped according to their associated variable or composite variable (Fig. 4a, b). SEM was performed using the "lavaan" package (Rosseel, 2012) in R (R Development Core Team, 2017). Providing specific details on the mathematics involved in the SEM method and the "lavaan" package is beyond the scope of this article. For a more detailed description of the SEM method and the "lavaan" package, see Bollen (1989), Grace and Bollen (2006, 2008), Rosseel (2012) and Shipley (2016). An initial analysis of the SEMs did not indicate a good model fit when including the composite variables "plants" and "earthworms" in a DAG simultaneously. Therefore, the influence of MADOWS, soil parameters, plants and earthworms on topsoil structure stability was tested in two separate models for the composite variables "plants" and "earthworms". The analysis of causal relations and path coefficients of soil parameters and MADOWS on topsoil structure stability is not restricted when running two separate models for the analysis of the influence of plants and earthworms.

3. Results

3.1. Topsoil structure stability and soil parameters

The mean percentage of water-stable aggregates was around 30% in the floodplain, but ranged between values of 8 and 62% (Table 2). The percentages of silt and sand, and the base saturation show high standard deviations indicating a large variability within the plots. Standard deviations for CEC, total and active carbonates, TOC contents and pH values do not show a large variability between the sampling plots (Table 2).

3.2. Mean annual days of water saturation (MADOWS)

Simulated mean water saturation in the topsoil, averaged over the simulated 4 years (2013–2016), ranged from 25 to 30%. Average annual values were slightly increased for 2013 and 2014 compared to 2015 and 2016 (data not shown). The mean annual days with complete water saturation (MADOWS100) was <10 averaged over all plots, and ranged between 0 and 25 days for the individual plots (Fig. 5). The mean number of days with partial water saturation tended to be increased stepwise by the factors 1.5 for MADOWS85 and MADOWS70, but was not statistically significant. The mean annual days >55% water saturation (MADOWS55) was significantly higher than for MADOWS100 ($P < 0.001$), but not for MADOWS85 and MADOWS70. MADOWS40 was almost 3 times higher compared to MADOWS55, but only differed significantly from MADOWS70, MADOWS85 and MADOWS100 ($P < 0.001$). MADOWS25 was >350 days for all plots during the entire modelling period and differed significantly from MADOWS100 – MADOWS40 ($P < 0.001$).

3.3. Ecosystem engineers

Mean values for root size and root abundance factors (see Section 2.3) were greater for herbs than for bushes (Table 2). Root size and root abundance factors indicated a strong variation for both herbs and bushes. Both factors could not be calculated for shrubs in three sampling plots, as no bushes were recorded in the corresponding relevees.

Mean earthworm abundance of all ecological categories was 157 individuals (ind) m^{-2}. Total abundances of earthworms showed a broad range of variation, ranging from 0 ind m^{-2} to >750 ind m^{-2}. Endogeics showed the highest mean abundance in all plots followed by anecics and epigeics. Epigeics and anecics were completely absent in several sampling plots, but contributed to almost 30% (epigeics) and 50% (anecics) to the total abundance of earthworms in others. Percentage of endogeics tended to be equally distributed in all sampling plots.

3.4. Structural equation modelling SEM

Full structural equation models could be fitted for complete and partial water saturation, as presented in Fig. 6a, b for MADOWS100. Path coefficients for partial water saturation are presented in Table 3. According to the SEM, the composite variables "soil parameters" and "plants" have a significant positive effect on the topsoil structure stability with path coefficients of 0.68 and 0.31. The silt content has the strongest positive impact on the composite variable "soil parameters" (path coefficient of 0.98), followed by CEC and TOC. Sand content and the pH value have strong negative impacts (path coefficients of −0.81 and

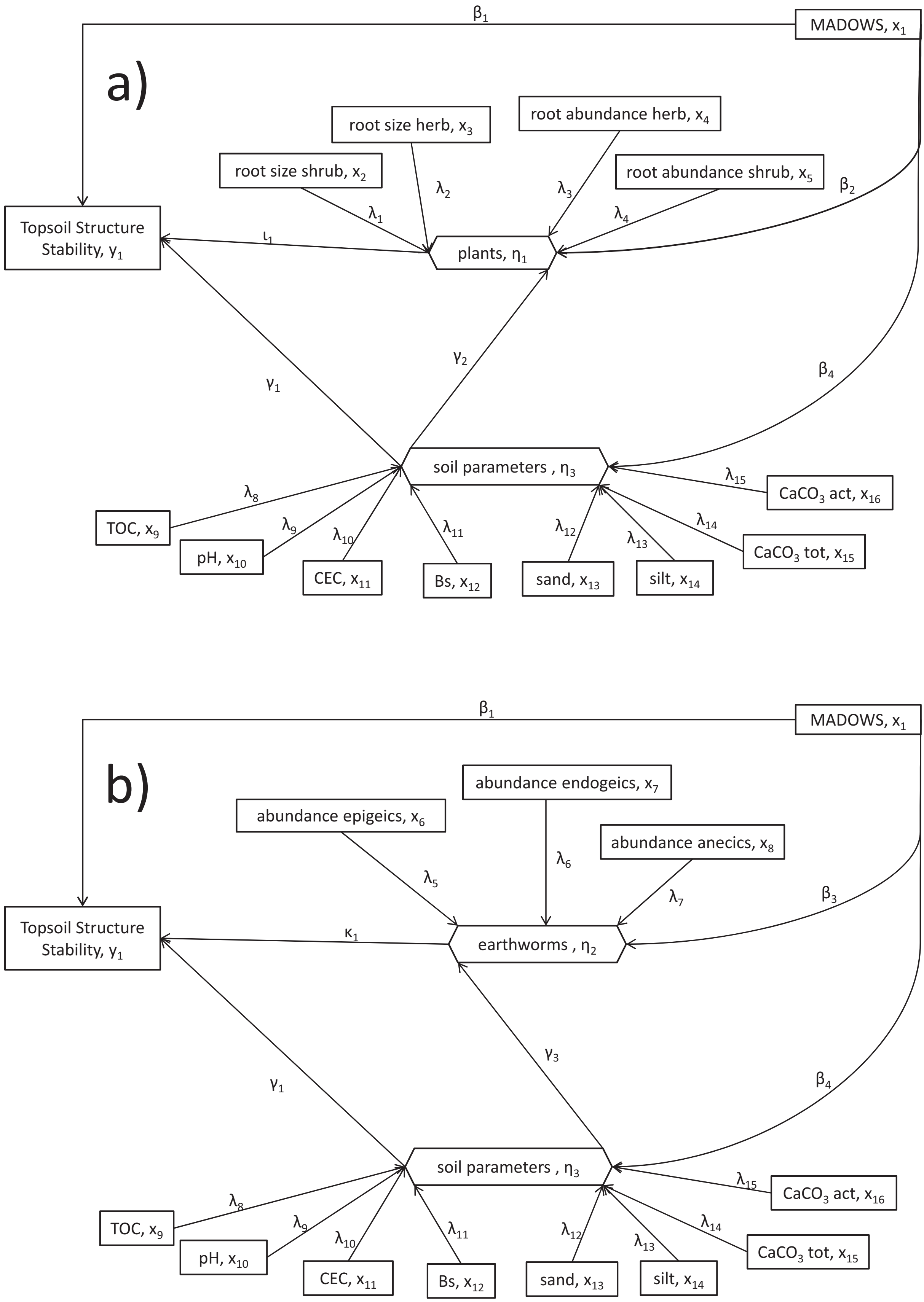


Fig. 4. The two directed acyclic graphs (DAGs) which were proposed as hypotheses for direct or indirect relations testing by structural equation modelling (SEM). The two DAGs differ by the composite variables a) "plants, η1" and b) "earthworms, η2". All observed variables are illustrated in boxes, while x illustrates the state as predictor variable and y as the response variable. Associated composite variables are bordered by a hexagonal shape and marked by η. Arrows mark pathways of hypothesised effects. Path coefficients are marked by β, γ, ι, κ and λ, and grouped according to predictor variables. MADOWS stands for mean annual days of water saturation, CEC for cation exchange capacity and Bs for base saturation.

Table 2
Descriptive statistical values for topsoil structure stability, plants, earthworms and soil parameters with n = 45 observations. "ind" stands for individuals and "CEC" for cation exchange capacity.

Parameters	Mean	min	max	Standard deviation
Water stable aggregates (%)	29.91	8.53	62.20	12.64
pH	7.97	7.53	8.35	0.18
Silt content (%)	31.54	3.80	71.10	22.95
Sand content (%)	63.16	27	94.10	23.65
CEC (cmolc kg^{-1})	16.36	8.31	29.62	4.87
Base saturation (%)	70.89	42.79	94.69	12.08
Total $CaCO_3$ (%)	28.94	27.10	30.09	0.64
Root size factor herbs	0.53	0.03	1.00	0.36
Root size factor shrubs	0.12	0.00	0.30	0.11
Root abundance factor herbs	0.62	0.09	1.75	0.47
Root abundance factor shrubs	0.29	0.00	2.40	0.75
Abundance epigeics (ind m^{-2})	10.21	0.00	123.80	27.47
Abundance endogeics (ind m^{-2})	73.66	0.00	503.04	114.00
Abundance anecics (ind m^{-2})	60.39	0.00	414.00	77.52

−0.40). The remaining observed soil physico-chemical properties did not contribute significantly to "soil parameters". The root size and root abundance factors for both shrubs and herbaceous plants have strong effects on the composite variable "plants", with positive path coefficients for shrubs and negative ones for herbs. No significant influence of the composite variable "earthworms" on the topsoil structure stability was found. While MADOWS100 did not show a significant direct influence on topsoil structure stability, it did on "plants" (path coefficient of 0.33) and on "soil parameters" (path coefficient of −0.50). MADOWS100 thus had an indirect effect on topsoil structure stability by positively influencing "plants" and negatively influencing "soil parameters", which in turn both showed a positive impact on topsoil structure stability. Significance levels of causal relations and values of path coefficients did not show large variations when running the model with MADOWS85, MADOWS70, MADOWS55 and MADOWS40 (Table 3). Significant effects of mean annual days of partial water saturation on "soil parameters" and "plants" disappeared only for MADOWS25, indicating that the average water saturation during a year (see Section 3.2) in the topsoil neither controlled topsoil structure stability nor explained occurrence of soil parameters and distribution patterns of ecosystem engineers.

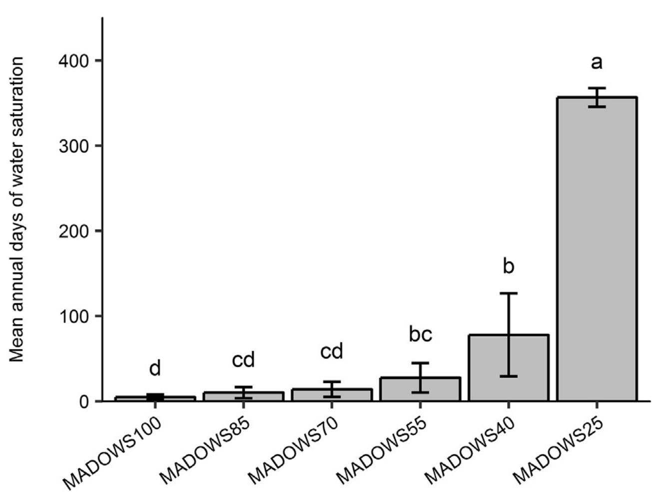


Fig. 5. Barplots showing modelled values for the mean annual days of water saturation (MADOWS) in the first 20 cm of soil for 100% (MADOWS100), for >85% (MADOWS85), for >70% (MADOWS70), for >55% (MADOWS55), for >40% (MADOWS40) and for >25% (MADOWS25) water saturation. Error bars represent standard deviations of values for n = 45 observations. Letters a, b, c and d represent results from Tukey's HSD tests for one-way ANOVA at significance level P < 0.001.

4. Discussion

4.1. Applicability of the SW-GW model

In many respects, the application of a SW–GW flow model was suitable to analyse the influence of fluctuating water levels on topsoil structure stability, ecosystem engineers and soil parameters. Over the simulated period (2013–2016), for example, two sampling plots were visually submerged only once in four years by a flood event and only considering SW would thus have led to the assumption that these plots are not regularly and often influenced by alluvial dynamics. However, based on our SW-GW flow simulations, it was revealed that in all sampling plots, GW often rose within the upper 20 cm of soil, which impacted ecosystem engineers similar to flooding by SW. The SW-GW simulations thus allowed reproducing the influence of fluctuating water levels much more holistically than pure flood observations or SW simulations, which would have clearly underestimated the effects of water level fluctuations on the topsoil structure stability. However, numerical SW-GW flow simulations are themselves subject of uncertainty due to the heterogeneous, and often unknown nature of the subsurface. In our current study, uncertainties in groundwater levels of up to 20 cm were observed, but the simulated water levels corresponded well with the observed ones during the crucial periods of low and average river water discharge. Higher uncertainties were observed during high and extreme discharge periods, e.g., during floods. As the water levels during high- and extreme-flow periods can rise up to 1 m above terrain, the investigated topsoil would be fully saturated even under consideration of 20 cm of uncertainty. The impacts of these flow model uncertainties are therefore minimal for the purpose of our analysis. Evapotranspiration was not included in the model simulation. In this particular configuration of boundary conditions and hydraulic properties, the explicit simulation of evapotranspiration would not have significantly changed the water table dynamics and thus the subsequent statistical analysis: there is a direct hydraulic connection between the floodplain and the river, so evapotranspired water would have been replaced by river water. Given that the hydraulic conductivity of the site is very high and the spatial scales small, evapotranspiration would immediately be compensated by the increased inflow from the river, thus having minimal influence on water table dynamics. The last major flood event at NNF in early 2013 dramatically changed the riverbed morphology, and information on the morphology of the floodplain before this event are not available. As the morphology of the river and the floodplain is a critical factor for flooding and for the exchange flow dynamics between SW and GW, sufficiently accurate flow simulations of years prior to 2013 could not be carried out. During the simulated period (2013–2016), only smaller flooding events occurred, which did not significantly modify the river morphology. For the simulated period, the SW-GW flow model was nevertheless an appropriate approach to determine more specifically groundwater level fluctuations in soil. To the best of our knowledge, this study is the first of its kind which analyses the relations between fluctuating groundwater levels, ecosystem engineers and soil parameters with regard to the stability of the topsoil structure.

4.2. Effect of fluctuating groundwater levels on topsoil structure stability

The results from the SEMs indicated that neither the number of days with complete, nor the number of days with partial water saturation in the upper 20 cm of soil directly influenced topsoil structure stability. However, both scenarios indicated strong influences on soil parameters and plants that, in turn, influence topsoil structure stability. Complete and partial water saturation thus showed no direct, but a strong indirect effect on topsoil structure stability. As reported by Bullinger-Weber et al. (2014), the percentage of water-stable macro-aggregates decreases with increasing flood frequency. However, in our study, a direct relation between SW-GW interaction and the topsoil structure stability

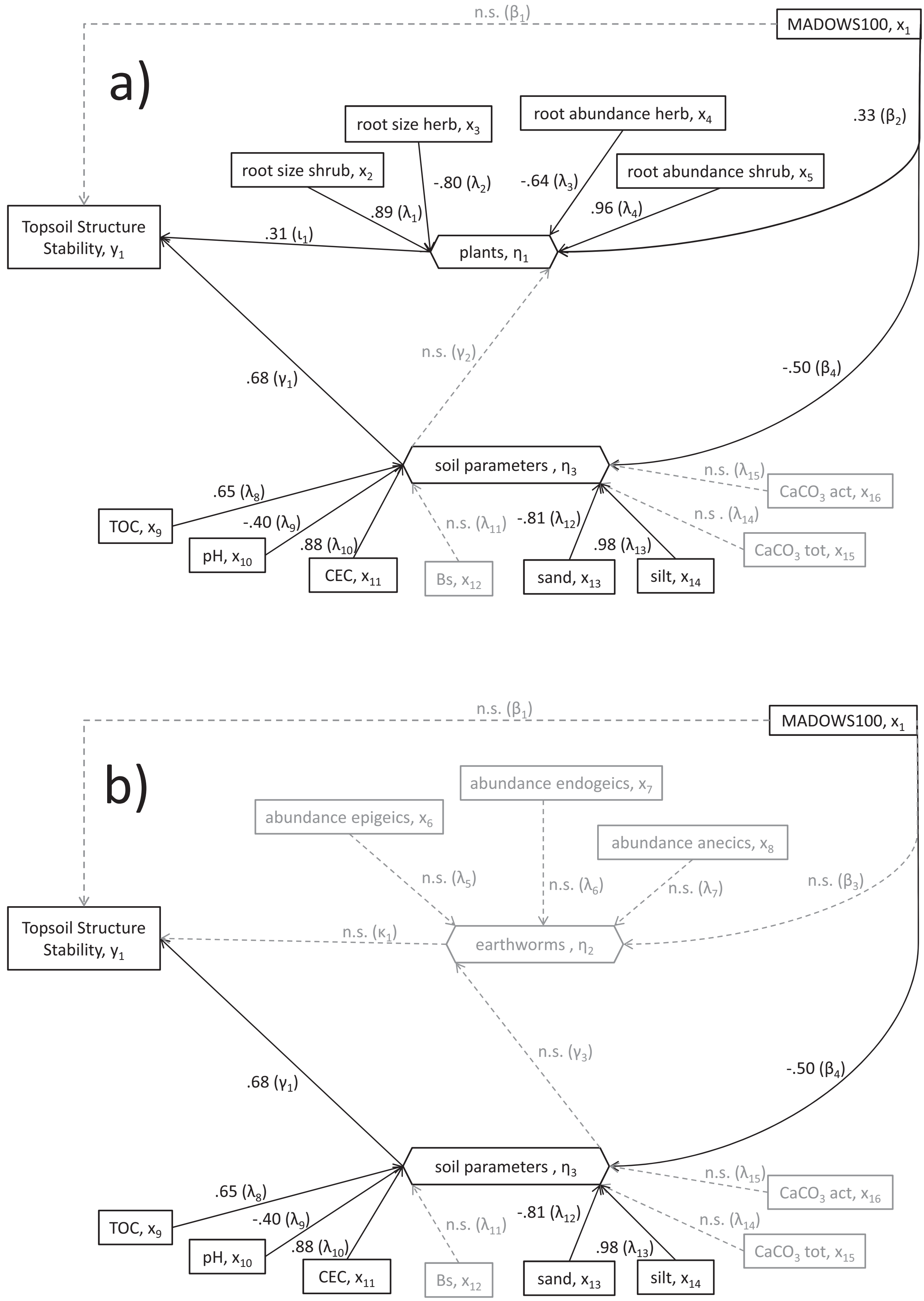


Fig. 6. Results obtained from SEM for mean annual days of water saturation (MADOWS) = 100% printed as directed acyclic graph (DAG) for a) plants and b) earthworms including path coefficients for significant relations at $P < 0.05$ level. Paths showing no significant effects (n.s.) are dashed and grey-coloured. Letters x and y represent vectors of observed predictor and response variables, η are vectors containing composite variables. Path coefficients are marked by β, γ, ι, κ and λ, and grouped according to predictor variables CEC stands for cation exchange capacity and Bs for base saturation.

Table 3
Values for path coefficients for partial water saturation in the topsoil. MADOWS stands for mean annual days of water saturation. Path coefficients correspond to the pathways presented in Fig. 2. Non-significant path coefficients are indicated by n.s.

Path coefficient	MADOWS				
	>85%	>70%	>55%	>40%	>25%
β_1	n.s.	n.s.	n.s.	n.s.	n.s.
β_2	0.43	0.47	0.44	0.47	n.s.
β_3	n.s.	n.s.	n.s.	n.s.	n.s.
β_4	−0.49	−0.43	−0.47	−0.48	n.s.
γ_1	0.63	0.64	0.63	0.65	0.47
γ_2	n.s.	n.s.	n.s.	n.s.	n.s.
γ_3	n.s.	n.s.	n.s.	n.s.	n.s.
ι_1	0.26	0.25	0.25	0.24	0.26
κ_1	n.s.	n.s.	n.s.	n.s.	n.s.

was not significant, neither for the number of days with complete, nor with partial water saturation. Floods usually have a destructive impact on soil structural stability, as scour forces at the soil surface lead to topsoil erosion (Cierjacks et al., 2011). In our study, the number of days with complete water saturation partly corresponding to floods was low during the modelled period and thus might not have affected the topsoil structure stability. Occurring more frequently than full inundations, GW table fluctuations partly saturating the topsoil did, however, not influence its structural stability either. The fluctuation of groundwater levels is smooth and physical forces do not have a destructive effect on soil structure stability compared to floods. Conversely, desiccation and remoistening of soil aggregates were even shown to improve their structural stability (Shipitalo and Protz, 1988; Decaëns, 2000; Shipitalo and Le Bayon, 2004). Fluctuating GW levels thus can also have an overall effect of improving structural stability in topsoil. Against our expectations, even a relatively low partial water saturation (i.e., MADOWS40) exerted a significant negative influence for soil parameters and positive influence for plants. The effect disappeared only at MADOWS25, which ranged slightly below the average water saturation of 25–30%. Low rates of water saturation were expected for the plot with sandy soils, but not for soils that contain larger proportions of finer soil texture. The floodplain soils thus appear to be efficiently drained, independent of the amount of finer soil particles, which allows them to rapidly dewater after a high-water event. This reasoning is supported by Liernur (2016) and Liernur et al. (2017), who, in the same sampling plots, measured extraordinary high water infiltration rates that were linked to a well-structured macro-porous system extending to a depth of 50 cm and thus favouring drainage capacity of the topsoil. This draining pathway likely explains the low mean contents of annual water saturation (i.e., below 30% for all sampling plots).

4.3. The effect of soil parameters on topsoil structure stability

According to the SEMs, topsoil structure stability was strongly influenced by soil parameters, mainly by the soil texture, pH and TOC values, as well as cation exchange capacities. Several studies have already shown a positive correlation between the fine soil fraction and the soil structure stability (Sollins et al., 1996; Bronik and Lal, 2005; Bullinger-Weber et al., 2007). On the other hand, aggregates composed of coarser material are considered less stable than those composed of finer material (Guenat et al., 1999; Kaiser et al., 2012). In the latter case, the interaction between the fine sand fraction and organic matter is of great importance for soil aggregate formation (Bronik and Lal, 2005). It is therefore not surprising that the TOC content showed a significant positive effect on soil aggregation in this study. Organic matter which is incorporated into soil aggregates chemically binds to mineral particles or is occluded into soil aggregates and improves the structural stability of soils (Tisdall and Oades, 1982; Chorom et al., 1994; Tisdall, 1996; Haynes and Naidu, 1998; Kong et al., 2005). The importance of pH values and cation exchange capacity found in this study has also been found in previous studies, whereby the effect on soil structure stability is rather indirect: increased pH values can enhance microbial activity and clay dispersion (Haynes and Naidu, 1998; Chorom et al., 1994), whereas cation exchange capacity can reduce repulsive forces of negatively charged fine soil particles (Tisdall, 1996). Calcium carbonates did not show any effect on soil aggregates in this study, although they usually act as a cementing agent which agglutinates soil particles (Bronik and Lal, 2005). The influence of carbonates was likely not significant as neither total nor active carbonate values indicated any variation between the sampling plots (data not shown). The majority of the soil physico-chemical parameters are mutually interrelated: for example, an increased silt content usually causes a higher cation exchange capacity compared to a sandy soil. This could be a reason for the extraordinary strong relation between MADOWS and soil parameters found in this study. Especially during flood events, alluvial sediments are deposited in the floodplain area, whose texture usually depends on the flow velocity (Cierjacks et al., 2011; Graf-Rosenfellner et al., 2016).

4.4. The effect of plants on topsoil structure stability

Root size and root abundance factors (see Section 2.3) showed a significant positive effect on topsoil structure stability, and were positively affected by MADOWS. “Herbs” had a negative effect on the composite “plants” in contrast to “shrubs”, whose effect was strongly positive. As already highlighted in several studies, plants highly contribute to soil aggregation through their root system and thus improve the structural stability in soils: on one hand, soil particles are physically enmeshed during the extension of the root system in the rhizosphere. On the other hand, plant roots release root exudates that directly agglutinate soil particles or stimulate rhizospheric microorganisms, improving the structural stability of soils (Degens et al., 1994; Angers and Caron, 1998; Blanchart et al., 2004; Fonte et al., 2012). However, MADOWS had a positive effect on “plants”, particularly on shrubs. Pioneer shrubs, such as willows, are well adapted to periodic water saturation and to recurring floods over their branched root system (Crouzy and Perona, 2012; Perona et al., 2012; Bätz et al., 2014). In close proximity to the river, plots were dominated by pioneer herbaceous plants whose root system is similar to those of shrubs in terms of functioning. However, SEM results indicated that the proportion of herbs decreased with increasing flood frequency. In plots more distant to the river, pioneer herbaceous plant species are replaced by other herbaceous plants assembled to an alluvial forest alliance. There, especially the abundance of shrubs decreases. This can be explained by the almost closed canopy in the post-pioneer riparian forest which established under a decreasing effect of flood frequency allowing a development of more stabilised habitats with less frequent disturbances (Corenblit et al., 2009).

4.5. The effect of earthworms on topsoil structure stability

In the SEMs, earthworm abundance surprisingly did not show any influence on topsoil structure stability and was not explained by MADOWS or soil parameters. These results are somewhat contradictory to the current state of the literature, as many earthworms, especially endogeics and anecics, are known to efficiently structure soils (Schäfer and Schauermann, 1990; Brown et al., 2000; Kong and Six, 2010) and show large dependencies on soil properties and variable soil moisture contents. Anecics are less tolerant to frequent disturbances and prefer habitats with more stable environmental conditions, larger soil depth and a finer soil texture (Guenat et al., 1999; Bullinger-Weber et al., 2012; Le Bayon et al., 2017). However, the abundance of anecics is not restricted to the upper 20 cm of soils in which the generally low water content increases the risk of desiccation. This fact might be one explanation why soil properties of the upper 20 cm did not have a significant influence on their abundance, and why the topsoil structure stability was not affected by anecics. Surprisingly similar, SEMs did not show any significant relations of endogeics and epigeics on topsoil structure stability or interferences through soil parameters and MADOWS either. Especially endogeics are considered to be very important for soil aggregation (Brown et al., 2000). A potential reason for the absence of a significant influence according to SEM could be the fact that the occurrence of endogeics did not show large variability between the sampling plots, which could be a result of their higher tolerance to varying environmental conditions (Ivask et al., 2007). Epigeics are to a lesser extent affected by soil properties because they do not get directly in touch with the soil. Epigeics rather live on top of the soil feeding on litter rather than rummage the soil matrix or contribute to topsoil structure stability (Shipitalo and Protz, 1988; Bullinger-Weber et al., 2007; Eijsackers, 2010; Salomé et al., 2011). Water saturated soils, in general, have a negative impact on earthworms' abundance (Plum and Filser, 2005; Zorn et al., 2008). However, the mean annual water saturation in the topsoil was only around 25–30%, and days with higher water saturations were low due to the efficient drainage of the soils (Liernur, 2016). The period during which earthworms were stressed by rising water levels in the soil over the measurement period was thus low over considering the course of a year. Furthermore, earthworms have developed efficient strategies to avoid water saturated conditions in soil or survive in water saturated soils for a short time: anecics try to escape the habitat as they are less tolerant to water saturated soils (Plum and Filser, 2005). Endogeics can tolerate water-saturated conditions up to several weeks through physiological adaptations: *A. chlorotica* and *A. caliginosa* for example stay in diapause by reducing metabolic activities during a flood period and recover as soon as a flood period has ended (Plum and Filser, 2005; Zorn et al., 2008). During water saturated conditions in soil, the population of epigeics almost entirely collapses, but recovers rapidly due to their high re-succession rate and their cocoons that can outlast water saturated conditions for several months (Roots, 1956; Plum and Filser, 2005). Water saturated conditions thus have a strong direct impact on abundance of epigeics – but this effect could not be reproduced by the SEMs due to their extraordinary high and rapid re-colonisation rate.

4.6. Scopes and limitations of SEM to our experimental design

The application of SEM was suitable for the analysis of direct relations and complex multiple interactions of different variables in this study. The indirect effect of fluctuating groundwater levels on topsoil structure stability over plants and soil parameters could be highlighted, but would have been remained concealed with regression modelling only. SEM was thus an appropriate way to analyse the data and visualise complex causal relations between the variables in our study. The relations of the composite variables “plants” and “earthworms” could not be implemented in the same SEMs due to the insufficient model fit, which resulted most likely from the limited number of samples. Equally a result of the low number of replicates, the predictor variables were not enlarged with additional parameters, such as the biomass of earthworms (Grace and Bollen, 2008; Shipley, 2016). Furthermore, the different time scales of the variables made it complicated to implement the variables in the same SEMs at first glance. The characteristic timescale associated with soil parameters, for example, are significantly longer compared to the population dynamics of earthworms. However, soil development in floodplains is a continuous process, as major flood events periodically burrow former topsoils and superpose them with thick layers of alluvial sediments. The last major flood event impacting all habitats dates back to 2013, shortly before the beginning of our simulation period. As only a handful of smaller flood events but no further major one influenced the habitats, it can be assumed that topsoil structure and parameter development sampled and analysed in this study has developed rather smoothly from 2013 on forward. The same applies to plants and earthworms that have recolonised the topsoil after this event. The relatively calm period between 2013 and 2016 has enabled the development of rather stable plant and earthworm populations that fluctuate between seasons and years, but have not completely collapsed since 2013. The populations at the moment of sampling thus reflected the result of community dynamics of the past years. Therefore, it can be considered as a decisive factor for the modelling outcome that all predictor variables were shifted to a similar time scale during the investigated time period. The SEM thus properly describes relations and functions for the simulated time period between 2013 and 2016. However, one cannot infer whether the found relations also hold true for a different period or another floodplain ecosystem with stronger or weaker alluvial dynamics. Nevertheless, our results strongly suggest that the found relations are assignable to similar alluvial systems unless soil development is reset or communities of ecosystem engineers collapse due to major flood events.

5. Conclusions

The combination of numerical flow simulations using a physically-based, fully-coupled SW-GW model with SEM allowed identifying causal relations and multiple interactions between the topsoil structure stability, ecosystem engineers, and soil properties under complete and partial water saturation of the topsoil. Neither complete nor partial water saturation had a direct influence on topsoil structure stability, but an indirect influence by affecting distribution patterns of plant roots and soil parameters that had, in turn, a significant effect on topsoil structure stability. The distribution of earthworm populations was neither significant for topsoil structure stability nor affected by soil parameters and fluctuating GW levels. Fluctuating GW levels thus control abiotic factors in the plots as well as the distribution patterns of plants and their capability to structure the topsoil in our study. However, earthworm populations are much more independent from fluctuating GW levels and abiotic conditions of their habitats compared to plants, as they developed adaptation strategies which allow them tolerating harsh environmental conditions. Analysing GW levels in soil was more appropriate than purely observing flood frequencies, as partial water saturation in soil already had an effect on soil properties, plants and indirectly on topsoil structure stability. These effects would have been underestimated if only complete water saturation of the soil had been considered. The modelled period between 2013 and 2016 was a convenient period to study topsoil parameters and ecosystem engineers, without any major flood event burrowing the topsoil or extinguishing populations of ecosystem engineers. Therefore, the question remains open whether our results are specific for the modelled period and the floodplain at NNF. However, it is very likely that the influences of the different predictor variables on topsoil structure stability as found in our study are also applicable to comparable near-natural or restored floodplain systems. It can generally be stated that analysing relations between SW-GW dynamics and soil engineering organisms in floodplains is of crucial importance to evaluate the maintenance of ecosystem services for improving management strategies for near-natural floodplains and the success of floodplain restoration projects.

Acknowledgements

This work was developed as part of the FloodSTRESS project funded by the Swiss National Science Foundation (SNF), project no. FN 315230_153460, and the CCES-project RECORD Catchment of the ETH domain. Oliver S. Schilling gratefully acknowledges the funding provided by the SNF grant P2NEP2_171985. The authors gratefully thank all co-workers for their strong support in the field and in the laboratory: Dirk Radny and Reto Britt for administrative support in the field, Dovydas Rutkauskas for soil sampling and description as well as for the soil analysis in the lab, Eric Torrejon for earthworm sampling, Florence Arlettaz and Théodore Hafen for earthworm determination, Magalì Matteodo for vegetation description, Elena Rossel for granulometry, Armelle Vallat for ICP analyses, Mégane Rohrer and Amandine Pillonel for CHN analysis and the Fibl institute in Frick for providing equipment for the aggregate stability analysis. Furthermore, we thank the two anonymous reviewers for their highly constructive comments.

References

Acreman, M.C., Riddington, R., Booker, D.J., 2003. Hydrological impacts of floodplain restoration: a case study of the River Cherwell, UK. Hydrol. Earth Syst. Sci. 7, 75–85.

Angers, D.A., Caron, J., 1998. Plant-induced changes in soil structure: processes and feedbacks. Biogeochemistry 42, 55–72.

Aquanty Inc., 2016. HydroGeoSphere. A Three-dimensional Numerical Model Describing Fully Integrated Subsurface and Surface Flow and Solute Transport (Waterloo, ON, Canada).

Ausden, M., Sutherland, W.J., James, R., 2001. The effects of flooding lowland wet grassland on soil macroinvertebrate prey of breeding wading birds. J. Appl. Ecol. 38, 320–338.

Banks, E.W., Brunner, P., Simmons, C.T., 2011. Vegetation controls on variably saturated processes between surface water and groundwater and their impact on the state of connection. Water Resour. Res. 47. https://doi.org/10.1029/2011WR010544.

Bätz, N., Verrecchia, E., Lane, S.N., 2014. Soil in braided rivers: an overlook component of braided river morphodynamics. In: Schleiss, et al. (Eds.), River Flow. Tylor & Francis Group, London (2014).

Bätz, N., Verrecchia, E.P., Lane, S.N., 2015. Organic matter processing and soil evolution in a braided river system. Catena 126, 86–97.

Blakemore, R.J., 2008. An Updated List of Valid, Invalid and Synonym Names of Criodrioidea (Criodrilidae) and Lumbricoidea (Annelida: Oligochaeta: Sparganophilidae, Ailoscolecidae, Hormogastridae, Lumbricidae, and Lutodrilidae). Yokohama National University.

Blanchart, E., Lavelle, P., Braudeau, E., Le Bissonais, Y., Valentin, C., 1997. Regulation of soil structure by geophagous earthworm activites in humid savannas of Côte d'Ivoire. Soil Biol. Biochem. 29, 431–439.

Blanchart, E., Albrecht, A., Brown, G., Decaens, T., Duboisset, A., Lavelle, P., Mariani, L., Roose, E., 2004. Effects of tropical endogeic earthworms on soil erosion. Agric. Ecosyst. Environ. 104, 303–315.

Bollen, K.A., 1989. Structural Equations With Variable Variables. John Wiley & Sons, New York.

Bouché, M., 1972. Lombriciens de France. Ecologie et systématique. INRA Publ.Institut National des Recherches Agriculturelles, Paris

Bouché, M., Aliaga, R., 1986. L'échantillonnage des lombriciens: une urgente nécessité. Déf. Vég. 242, 30–36.

Braun-Blanquet, J., 1964. Pflanzensoziologie, Grundzüge der Vegetationskunde. 3. Auflage. Springer Verlag, Wien (865 pages).

Bronik, C.J., Lal, R., 2005. Soil structure and management: a review. Geoderma 124, 3–22.

Brooks, A., Shields, F.D., 1996. River Channel Restoration: Guiding Principles for Sustainable Restoration Projects. Wiley, Chichester, UK.

Brown, G.G., Barois, I., Lavelle, P., 2000. Regulation of soil organic matter dynamics and microbial activity in the drilosphere and the role of interactions with other edaphic functional domains. Eur. J. Soil Biol. 36, 177–198.

Brunner, P., Simmons, C.T., 2011. HydroGeoSphere: a fully integrated, physically based hydrological model. Ground Water 50 (2), 170–176.

Brunner, P., Simmons, C.T., Cook, P.G., 2009. Spatial and temporal aspects of the transition from connection to disconnection between rivers, lakes and groundwater. J. Hydrol. 376 (1–2), 159–169.

Brunner, P., Therrien, R., Renard, P., Simmons, C.T., Hendricks Franssen, H.J., 2017. Advances in understanding river-groundwater interactions. Rev. Geophys. https://doi.org/10.1002/2017RG000556.

Bullinger-Weber, G., Le Bayon, R.-C., Guenat, C., Gobat, J.-M., 2007. Influence of some physicochemical and biological parameters in soil structure formation in alluvial soils. Eur. J. Soil Biol. 43, 57–70.

Bullinger-Weber, G., Guenat, C., Gobat, J.-M., Le Bayon, R.-C., 2012. Impact of flood deposits on earthworm communities in alder forests from a subalpine floodplain (Kandersteg, Switzerland). Eur. J. Soil Biol. 49, 5–11.

Bullinger-Weber, G., Le Bayon, R.-C., Thébault, A., Schlaepfer, R., Guenat, C., 2014. Carbon storage and soil organic matter stabilisation in near-natural, restored and embanked Swiss floodplains. Geoderma 228, 122–131.

Chaudhary, V.B., Bowker, M.A., O'Dell, T.E., Grace, J.B., Redman, A.E., Rillig, M.C., Johnson, N.C., 2009. Untangling the biological contributions to soil stability in semiarid shrublands. Ecol. Appl. 19 (1), 110–122.

Chorom, M., Regasamy, P., Murray, R.S., 1994. Clay dispersion as influenced by pH and net particle charge of sodic soils. Aust. J. Soil Res. 32, 1243–1252.

Cierjacks, A., Kleinschmit, B., Kovarik, I., Graf, M., Lang, F., 2011. Organic matter distribution in floodplains can be predicted using spatial and vegetation structure data. River Res. Appl. 27, 1048–1057.

Corenblit, D., Steiger, J., Gurnell, A.M., Tabacchi, E., Roques, L., 2009. Control of sediment dynamics by vegetation as a key function driving biogeomorphic succession within fluvial corridors. Earth Surf. Process. Landf. 34, 1790–1810.

Crouzy, B., Perona, P., 2012. Biomass selection by floods and related timescales. Part 2: stochastic modeling. Adv. Water Resour. 39, 97–105.

Davis, C.A., Austin, J.E., Buhl, D.A., 2006. Factors influencing soil invertebrate communities in riparian grasslands of the central Platte River floodplain. Wetlands 26, 438–454.

Decaëns, T., 2000. Degradation dynamics of surface cats in grasslands of the eastern plains of Colombia. Biol. Fertil. Soils 23, 149–156.

Degens, B.P., Sparling, G.P., Abbott, L.K., 1994. The contribution from hyphae, roots and organic carbon constituents to the aggregation of a sandy loam under long-term clover-based and grass pastures. Eur. J. Soil Sci. 45, 459–468.

Diaz-Zorita, M., Perfect, E., Grove, J., 2002. Disruptive methods for assessing soil structure. Soil Tillage Res. 64, 3–22.

Diem, S., Renard, P., Schirmer, M., 2014. Assessing the effect of different river water level interpolation schemes on modeled groundwater residence times. J. Hydrol. 510, 393–402.

Drouineau, G., 1942. Dossage rapide du calcaire actif du sol: nouvelles données sur la séparation et la nature des fractions calcaires. Annu. Agron. 12, 441–450.

Eijsackers, H., 2010. Earthworms as colonisers: primary colonization of contaminated land, and sediment and soil waste deposits. Sci. Total Environ. 408, 1759–1769.

Feurer, D., Bailly, J.-S., Puech, C., Le Coarer, Y., Viau, A.A., 2008. Very high-resolution mapping of river-immersed topography by remote sensing. Prog. Phys. Geogr. 32 (4), 403–419.

Fonte, S.J., Quintero, D.C., Velasquez, E., Lavelle, P., 2012. Interactive effects of plants and earthworms on the physical stabilization of soil organic matter in aggregates. Plant and Soil 359, 205–214.

Fournier, B., Samaritani, E., Shrestha, E., Mitchell, E.A.D., Le Bayon, R.-C., 2012. Community ecology of earthworm in a restored floodplain and potential as bioindicators of river restoration. Appl. Soil Ecol. 59, 87–95.

Fournier, B., Guenat, C., Bullinger-Weber, G., Mitchell, E.A.D., 2013. Spatio-temporal heterogeneity of riparian soil morphology in a restored floodplain. Hydrol. Earth Syst. Sci. 17, 4031–4042.

Fournier, B., Gillet, F., Le Bayon, R.-C., Mitchell, E.A.D., Moretti, M., 2015. Functional responses of multitaxa communities to disturbance and stress gradients in a restored floodplain. J. Appl. Ecol. 52, 1364–1373.

Gianni, G., Richon, J., Perrochet, P., Vogel, A., Brunner, P., 2016. Rapid identification of transience in streambed conductance by inversion of floodwave responses. Water Resour. Res. 52, 2647–2658.

Grace, J.B., Bollen, K.A., 2006. The Interface Between Theory and Data in Structural Equation Models. U.S. Geological Survey, Reston, Virginia.

Grace, J.B., Bollen, K.A., 2008. Representing general theoretical concepts in structural equation models: the role of composite variables. Environ. Ecol. Stat. 15, 191–213.

Graf-Rosenfellner, M., Cierjacks, A., Kleinschmit, B., Lang, F., 2016. Soil formation and its implications for stabilization of soil organic matter in the riparian zone. Catena 139, 9–18.

Guenat, C., Bureau, F., Weber, G., Toutain, F., 1999. Initial stages of soil formation in a riparian zone: importance of biological agents and lithogenic inheritance in the development of the soil structure. Eur. J. Soil Biol. 35, 153–161.

Gurnell, A., Petts, G., 2002. Island-dominated landscapes of large floodplain rivers, a European perspective. Freshw. Biol. 47, 581–600.

Gurnell, A., Petts, G., 2006. Trees as riparian engineers: the Tagliamento River, Italy. Earth Surf. Process. Landf. 31, 1558–1575.

Haynes, R.J., Naidu, R., 1998. Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: a review. Nutr. Cycl. Agroecosyst. 51, 123–137.

Henszey, R.J., Pfeiffer, K., Keough, J.R., 2004. Linking surface- and ground-water levels to riparian grassland species along the Platte River in central Nebraska, USA. Wetlands 24, 665–687.

Horat & Scherrer AG Hydrologie und Hochwasserschutz, 1999. Hochwasserschutz und Auenlandschaft Thur-Mündung: Hydrologische Grundlagen. Auftraggeber. Amt für Abfall, Wasser, Energie und Luft (AWEL), Kt. Zürich.

IUSS Working Group WRB, 2006. Guidelines for Soil Description. FAO, Rome.

IUSS Working Group WRB, 2015. World Reference Base for Soil Resources 2014, update 2015. International soil classification system for naming soils and creating legends for soil maps. World Soil Resources Reports No. 106. FAO, Rome.

Ivask, M., Truu, J., Kuu, A., Truu, M., Leito, A., 2007. Earthworm communities of flooded grasslands in Matsalu, Estonia. Eur. J. Soil Biol. 43, 71–76.

Junk, W.J., Welcomme, R.L., 1990. Floodplains. Wetlands and Shallow Continental Water Bodies vol. 1, pp. 491–524.

Kaiser, M., Ellerbrock, R.H., Wulf, M., Dultz, S., Hierath, C., Sommer, M., 2012. The influence of mineral characteristics on organic matter content, composition, and stability of topsoils under long-term arable and forest land use. J. Geophys. Res. Biogeosci. 117, 1–16.

Karan, S., Engesgaard, P., Rasmussen, J., 2014. Dynamic streambed fluxes during rainfall-runoff events. Water Resour. Res. 50, 2293–2311.

Käser, D., Graf, T., Cochand, F., McLaren, R.G., Therrien, R., Brunner, P., 2014. Channel representation in physically based models coupling groundwater and surface water: pitfalls and how to avoid them. Ground Water 52 (6), 827–836.

Kemper, W.D., Rosenau, R.C., 1986. Aggregate stability and size distribution. In: K. e. American Society of Agronomy - Soil Science Society of America (Ed.), Methods of Soil Analysis, Part 1. Agronomy Monograph No 9. ASA and SSSA, Madison, WI, pp. 425–442.

Kong, A.Y.Y., Six, J., 2010. Tracing roots vs. residue carbon into soils from conventional and alternative cropping systems. Soil Sci. Soc. Am. J. 74, 1201–1210.

Kong, A.Y.Y., Six, J., Bryant, D.C., Denison, R.F., van Kessel, C., 2005. The relationship between carbon input, aggregation, and soil organic carbon stabilization in sustainable cropping systems. Soil Sci. Soc. Am. J. 69, 1078–1085.

Kurtz, W., Lapin, A., Schilling, O.S., Tang, Q., Schiller, E., Baun, T., Hunkeler, D., Vereecken, H., Sudicky, E., Kropf, P., Hendricks Franssen, H.J., Brunner, P., 2017. Integrating hydrological modeling, data assimilation and cloud computing for real-time management of water resources. Environ. Model Softw. 93, 418–435.

Lavelle, P., Spain, A., 2001. Soil Ecology. Springer, Netherlands https://doi.org/10.1007/0-306-48162-6.

Lavelle, P., Bignell, D., Lepage, M., Wolters, V., Roger, P., Ineson, P., Heal, O.W., Dhillion, S., 1997. Soil function in a changing world: the role of invertebrate ecosystem engineers. Eur. J. Soil Biol. 33, 159–193.

Lawrence, A.P., Bowers, M.A., 2002. A test of the hot mustard extraction method of sampling earthworms. Soil Biol. Biochem. 34, 549–552.

Le Bayon, R.C., Bullinger-Weber, G., Schomburg, A., Turberg, P., Schlaepfer, R., Guenat, C., 2017. Earthworms as ecosystem engineers: a review. Earthworms: Types, Roles and Research. NOVA Science Publishers, New York, pp. 129–178.

Li, Q., Unger, A.J.A., Sudicky, E.A., Kassenaar, D., Wexler, E.J., Shikaze, S., 2008. Simulating the multi-seasonal response of a large-scale watershed with a 3D physically-based hydrologic model. J. Hydrol. 357, 317–336.

Liernur, A., 2016. Impact of River Restoration (Thur River, TG, Switzerland) in the Formation of Soil Structure Along an Evolutive Sequence – Contribution of X-ray Computed Tomography. EPFL, Lausanne (Master Thesis).
Liernur, A., Schomburg, A., Turberg, P., Guenat, C., Le Bayon, R.C., Brunner, P., 2017. Coupling X-ray computed tomography and freeze-coring for the analysis of fine-grained low-cohesive soils. Geoderma 308, 171–186.
Malmqvist, B., Rundle, S., 2002. Threats to the running water ecosystems of the world. Environ. Conserv. 29, 134–153.
McLaren, R.G., 2011. GridBuilder - A Preprocessor for 2-D, Triangular Element, Finite-element Programs. Edited, Groundwater Simulations Group, University of Waterloo, Waterloo, Ontario, Canada.
Murer, E.J., Baumgarten, A., Eder, G., Gerzabek, M.H., Kandeler, E., Rampazzo, N., 1993. An improved sieving machine for estimation of soil aggregate stability (SAS). Geoderma 56, 539–547.
Partington, D., Therrien, R., Simmons, C.T., Brunner, P., 2017. Blueprint for a coupled model of sedimentology, hydrology, and hydrogeology in streambeds. Rev. Geophys. 55 (2), 287–309.
Perona, P., Molnar, P., Crouzy, B., Perucca, E., Jiang, Z., MeLelland, S., Wüthrich, D., Edmaier, K., Francis, R., Camporeale, C., Gurnell, A., 2012. Biomass selection by floods and related timescales: part 1. Experimental observations. Adv. Water Resour. 39, 85–96.
Plum, M., Filser, J., 2005. Floods and drought: response of earthworms and potworms (Oligochaeta: Lumbricidae, Enchytraeidae) to hydrological extremes in wet grassland. Pedobiologia 49, 443–453.
R Development Core Team, 2017. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
Rillig, M.C., Wright, S.F., Eviner, V.T., 2002. The role of arbuscular mycorrhizal fungi and glomalin in soil aggregation: comparing effects of five plant species. Plant Soil 238, 325–333.
Roots, B.I., 1956. The water relations of earthworms II: resistance to desiccation and immersion, and behavior when submerged and when allowed a choice of environment. J. Exp. Biol. 33, 29–44.
Rosseel, Y., 2012. Lavaan: an R package for structural equation modeling. J. Stat. Softw. 48 (2), 1–36.
Salomé, C., Guenat, C., Bullinger-Weber, G., Gobat, J.-M., Le Bayon, R.-C., 2011. Earthworm communities in alluvial forests: influence of altitude, vegetation stages and soil parameters. Pedobiologia 54, 89–98.
Samaritani, E., Shrestha, J., Fournier, B., Frossard, E., Gillet, F., Guenat, C., Niklaus, P.A., Tockner, K., Mitchell, E.A.D., Luster, J., 2011. Heterogeneity of soil carbon pools and fluxes in a channelized and a restored floodplain section (Thur River, Switzerland). Hydrol. Earth Syst. Sci. Discuss. 8, 1059–1091.
Schäfer, M., Schauermann, J., 1990. The soil fauna of beech forests: comparison between a mull and a moder soil. Pedobiologia 34 (5), 299–314.
Schilling, O.S., Doherty, J., Kinzelbach, W., Wang, H., Yang, P.N., Brunner, P., 2014. Using tree ring data as a proxy for transpiration to reduce predictive uncertainty of a model simulating groundwater–surface water–vegetation interactions. J. Hydrol. 519, 2258–2271.
Schilling, O.S., Gerber, C., Partington, D.J., Purtschert, R., Brennwald, M.S., Kipfer, R., Hunkeler, D., Brunner, P., 2017a. Advancing physically-based flow simulations of alluvial systems through atmospheric noble gases and the novel ^{37}Ar tracer method. Water Resour. Res. 53, 10465–10490.
Schilling, O.S., Irvine, D.J., Hendricks Franssen, H.J., Brunner, P., 2017b. Estimating the spatial extent of unsaturated zones in heterogeneous river-aquifer systems. Water Resour. Res. 53, 10583–10602.
Schirmer, M., Luster, J., Linde, N., Perona, P., Mitchell, E.A.D., Barry, D.A., Hollender, J., Cirpka, O.A., Schneider, P., Vogt, T., Radny, D., Durisch-Kaiser, E., 2014. Morphological, hydrological, biogeochemical and ecological changes and challenges in river restoration – the Thur River case study. Hydrol. Earth Syst. Sci. 18, 2449–2462.
Shipitalo, M.J., Le Bayon, R.C., 2004. Quantifying the effects of earthworms on soil aggregation and porosity. In: Edwards, C.A. (Ed.), Earthworm Ecology, 2nd edition CRC Press, Boca Raton, pp. 183–200.
Shipitalo, M.J., Protz, R., 1988. Factors influencing the dispersibility of clay in worm casts. J. Am. Soil Sci. Soc. 52, 764–769.
Shipley, B., 2016. Cause and Correlation in Biology. A User's Guide to Path Analysis, Structural Equations and Causal Inference With R. second edition. Cambridge University Press.
Six, J., Elliott, E.T., Paustian, K., 2000. Soil structure and soil organic matter: II. A normalized stability index and the effect of mineralogy. Soil Sci. Soc. Am. J. 64, 1042–1049.
Six, J., Bossuyt, H., De Gryze, S., Denef, K., 2004. A history of research on the link between (micro) aggregates, soil biota, and soil organic matter dynamics. Soil Tillage Res. 79, 7–31.
Sollins, P., Homann, P., Caldwell, B.A., 1996. Stabilization and destabilization of soil organic matter: mechanisms and controls. Geoderma 74, 65–105.
Tang, Q., Kurtz, W., Brunner, P., Vereecken, H., Hendricks Franssen, H.J., 2015. Characterisation of river–aquifer exchange fluxes: the role of spatial patterns of riverbed hydraulic conductivities. J. Hydrol. 531, 111–123.
Tang, Q., Kurtz, W., Schilling, O.S., Brunner, P., Vereecken, H., Hendricks Franssen, H.J., 2017. The influence of riverbed heterogeneity patterns on river-aquifer exchange fluxes under different connection regimes. J. Hydrol. 554, 383–396.
Tanner, C.C., 2001. Plants as ecosystem engineers in subsurface-flow treatment wetlands. Water Sci. Technol. 44, 9–17.
Therrien, R., McLaren, R.G., Sudicky, E.A., Panday, S., 2010. HydroGeoSphere: a three-dimensional numerical model describing fully-integrated subsurface and surface flow and solute transport. Hydrogeosphere Manual (edited).
Thonon, I., Klok, C., 2007. Impact of a changed inundation regime caused by climatic change and floodplain rehabilitation on population viability of earthworms in a lowerRiver Rhine floodplain. Sci. Total Environ. 372, 585–594.
Tisdall, J.M., 1996. Formation of soil aggregates and accumulation of soil organic matter. In: Carter, M.R., Stewart, B.A. (Eds.), Structure and Organic Matter Storage in Agricultural Soils. CRC Press, Boca Raton, FL, pp. 57–96.
Tisdall, J.M., Oades, J.M., 1982. Organic matter and water-stable aggregates in soils. J. Soil Sci. 33, 141–163.
Vatan, A., 1967. Manuel de sédimentologie. Technip, Paris.
Vogt, T., Hoehn, E., Schneider, P., Freund, A., Schirmer, M., Cirpka, O.A., 2009. Fluctuations of electrical conductivity as a natural tracer for bank filtration in a losing stream. Adv. Water Resour. 33, 1296–1308.
Vogt, T., Schneider, P., Hahn-Woernle, L., Cirpka, O.H., 2010. Estimation of seepage rates in a losing stream by means of fiber-optic high-resolution vertical temperature profiling. J. Hydrol. 380, 154–164.
Wever, L.A., Lysyk, T.J., Clapperton, M.J., 2001. The influence of soil moisture and temperature on the survival, aestivation, growth and development of juvenile Aporrectodea tuberculate (Eisen) (Lumbricidae). Pedobiologia 45, 121–133.
Winter, T.C., Harvey, J.W., Franke, O.L., Alley, W.M., 1998. Ground Water and Surface Water: A Single Resource. USGS, Denver, USA.
Zorn, M.I., van Gerstel, C.A.M., Eijsackers, H., 2005. Species-specific earthworm population responses in relation to flooding dynamics in a Dutsch floodplain soil. Pedobiologia 49, 189–198.
Zorn, M.I., van Gerstel, C.A.M., Morrien, E., Wagenaar, M., Eijsackers, H., 2008. Flooding responses of three earthworm species, *Allolobophora chlorotica*, *Aporrectodea caliginosa* and *Lumbricus rubellus*, in a laboratory-controlled environment. Soil Biol. Biochem. 40, 587–593.