



HAL
open science

Soil transfer impacts restored soil profiles and hydrodynamic properties

Adeline Bulot, Elise Bourru, Stéphane Ruy, Thierry Dutoit

► **To cite this version:**

Adeline Bulot, Elise Bourru, Stéphane Ruy, Thierry Dutoit. Soil transfer impacts restored soil profiles and hydrodynamic properties. *CATENA*, 2023, 231, pp.1-15. 10.1016/j.catena.2023.107308 . hal-04143207

HAL Id: hal-04143207

<https://hal.science/hal-04143207>

Submitted on 27 Jun 2023

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.



Distributed under a Creative Commons Attribution - NonCommercial - NoDerivatives 4.0 International License

Soil transfer impacts restored soil profiles and hydrodynamic properties

Adeline Bulot^{a,d,*}, Elise Bourru^{b,c}, Stéphane Ruy^c, Thierry Dutoit^d

^a BAGAP, Institut Agro, INRAE, ESA, 49000 Angers, France

^b Syndicat Mixte du Bassin de Thau – 328, Quai des Moulins, 34200 Sète, France

^c EMMAH, Avignon Université, INRAE, 84914 Avignon, France

^d IMBE, Avignon Université, Aix Marseille Université, CNRS, IRD 84911 Avignon, France

ARTICLE INFO

Keywords:

Dry grassland
Ecological restoration
Haplic Cambisol
Reconstituted technosol
Soil hydrological properties

ABSTRACT

Natural and semi-natural grasslands, particularly dry grasslands, are high-value biodiversity hotspots in Europe and major challenges for ecological conservation and restoration. Soil transfer has been used successfully to restore dry grassland ecosystems (soil and vegetation), but should only be applied as a last resort and where an undegraded ecosystem has previously independently been earmarked for destruction.

Assessing overall restoration success means taking soil recovery into account as well as vegetation regeneration. In Mediterranean regions and under global warming, soil hydrodynamic properties are the most critical limiting factors for the restoration of dry ecosystems. This study analyses soil physico-chemical parameters and hydrodynamic properties in reconstructed soils after soil transfer following a petrol land spill. Six years after soil transfer and compared to the surrounding soil reference steppe, the horizons of the soil transfer treatments were significantly less colonised by plant roots and showed very few biological activities and earthworm galleries. Then, their porosity was also significantly lower. Hydrodynamic properties continue to differ significantly between the restored site and the surrounding reference steppe ecosystem. There is also less available soil water and water drains faster in all soil transfer treatments. This can be explained by significant higher finer grain size (clays, silt), chemical elements (N, P, K), and plant cover for soil transfer treatments but significant, lower levels of organic matter due to mechanical soil compaction during soil spreading. The soil transfer treatment involving full reconstitution of the three main soil layers yields the most encouraging results. However, the question arises of whether long-term vegetation recovery might be compromised by soil hydrological dysfunction under global warming.

1. Introduction

In Western Europe, dry grasslands are often qualified as hotspots of biodiversity (Alard and Poudevigne, 2002; Gibson and Gibson, 2008; Habel et al., 2013; Poschlod and WallisDeVries, 2002; Weiher and Keddy, 1999; Willems, 2001). However, since the second half of the 20th century, there has been a significant loss of these dry grasslands (Adriaens et al., 2006; Buisson et al., 2004; Dutoit et al., 2013; Dzwonko and Loster, 1998; Lee et al., 2001) due to pressure from the growing human population (Gibson and Gibson, 2008; Lee et al., 2001; White, 2000). Particularly strong pressure has come from land-use changes such as agricultural intensification or abandonment, urbanisation or industrialisation (Balmford and Bond, 2005; Queiroz et al., 2014; Williams et al., 2009). This is especially true of the Mediterranean region, whose dry grasslands need to be both preserved and restored (Bonet,

2004; Buisson et al., 2021; Janišová et al., 2011; Traba et al., 2003).

Among the restoration methods successfully tested, soil transfer stands out for its excellent results in restoring ecosystems (Clewell and Aronson, 2013). Soil transfer involves extracting soil and biota from a donor site previously scheduled for destruction and transferring them to a receiver site, often at a ratio of 1:1 (Bulot et al., 2017; Kiehl et al., 2010). This method ensures the best restoration of soil physical and biological characteristics (Lamb et al., 2015), and plays an important role in the restoration of the plant community (Bulot et al., 2014b; Török et al., 2011; Wubs et al., 2016). Seeds and other propagules, such as rhizomes, small shallow-rooted plants and microorganisms, are transferred together with soil nutrients and materials (Buisson et al., 2018; Golos et al., 2016; Harris et al., 2006; Jasper, 2007; Reynolds et al., 2003; Török et al., 2011; Wubs et al., 2016). This is particularly effective when the topsoil layer translocation method, also called bulk topsoil

transfer or sod dumping, is used (Buisson et al., 2018; Bullock, 1998; Bulot et al., 2017, 2014b; Jaunatre et al., 2014; Kiehl et al., 2010; Piqueray et al., 2020). An alternative for severely degraded ecosystems may be direct transfer or hauling, which preserve the properties and soil quality of the donor site. Moreover, direct transfer can avoid or minimise seed loss or microbial biomass reduction at the receiver site (Anderson et al., 2008; DePuit, 1984). However, soil transfer is not a sustainable restoration process and should be used as a last resort: soil, particularly topsoil, is a rare and non-renewable resource that takes centuries to create, and its preservation comes before its restoration.

Several studies focusing on the importance of topsoil in rehabilitation have examined short and longer-term changes in nutrient availability, organic matter composition, microbial activity or density of viable plant seed stocks (Bulot et al., 2014b; Chenot et al., 2017, 2018). Their findings point to the need to consider the soil as well as vegetation recovery to better measure overall restoration success (Chenot et al., 2017, 2018; De Deyn et al., 2003; Piqueray et al., 2011). For example, adequate levels of soil physicochemical properties such as particle size, pH, electrical conductivity, organic carbon, nitrogen and other nutrients associated with soil fertility are crucial for the long-term survival of targeted plant communities (Alday et al., 2012; Frouz et al., 2008; Piqueray et al., 2011).

Among these soil properties, soil water content is one of the most critical limiting factors in the regeneration and distribution of vegetation in dry environments (Enright et al., 2005; McKenzie et al., 2004; Pueyo et al., 2013; Wu et al., 2014). In these arid and semi-arid regions, soil weathering is exacerbated by unpredictable rainfall and periods of drought, which can drastically limit vegetation recovery after soil reconstruction. An enhanced understanding of these hydrological processes, particularly the dynamics of water in reconstructed soils, is therefore essential (Audet et al., 2013; Beven and Germann, 2013). The water infiltration rate and hydraulic conductivity of vegetated soil are two important parameters governing surface runoff, soil moisture or the recharge of the groundwater table. It is particularly important to quantify these parameters, which allow rainfall capture and infiltration, together with water retention capacity, when restoring the soil of water-limited ecosystems. These limiting factors are useful indicators of soil degradation and drought potential (Ludwig et al., 2005; Michaelides et al., 2009; Sheoran et al., 2010). Soil physical parameters also need to be taken into account because they can influence hydrological processes. For example, soil water retention capacity is linked to soil particle-size distribution or soil texture (Khlosi et al., 2013; McBride and Mackintosh, 1984; Wösten et al., 2001).

In the Mediterranean region, where an increasingly dry and windy climate due to climate changes can limit plants' germination, growth or survival, quantifying soil physical and hydrological properties is vital to measure soil restoration success (Harris et al., 2006; Weng and Luo, 2008). Here, we conducted a field experiment to measure these properties in a soil transfer restoration operation that followed a petrol land spill in August 2009 in the plain of La Crau in southeastern France. We expected the results to shed light on the relative importance of soil parameters, including hydrological properties, in the success or failure of restoration operations involving soil transfers.

In 2010, the polluted soil was removed and a soil transfer was subsequently performed to enhance the regeneration of the typical pre-existing Mediterranean steppe (Bulot et al., 2014b). The soil transfer was carried out in 2011 to test the importance of faithfully reconstituting the vertical organisation of the three main soil layers (topsoil, subsoil, altered bedrock) as a reconstituted technosol (see Bulot et al., 2017 for more details of the soil transfer). The impacts of this restoration operation on the morphology of the soil profile and soil physical-chemical properties were measured in 2011, 2013 and again in 2016. Its impacts on soil hydrodynamic properties were also measured for the first time in 2016, to determine the medium-term success of physical soil recovery (6 years after soil transfer).

The main objective was to characterise and to compare the hydraulic

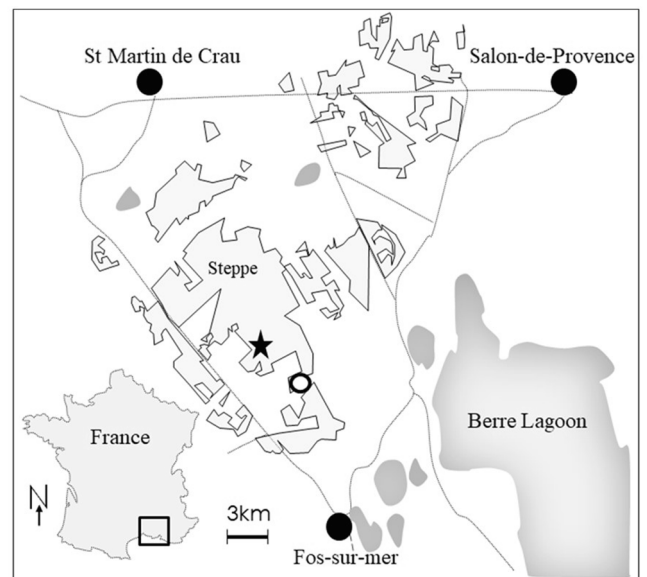


Fig. 1. Location of the plain of La Crau in France (black square), and location of study site (black star) and soil donor site (white circle) in the pseudo-steppe vegetation of the plain of La Crau (grey areas).

and physico-chemical properties of a reconstructed soil six years after its transfer. The spring 2011 restoration respected the vertical organisation of the different pedological layers of the steppe ecosystem's pre-existing natural Haplic Cambisol before its destruction by a petrol leak (2009). The objective was to restore the same typical Mediterranean steppe soil and its vegetation.

We assumed that (1) reconstituting the main pedological layers, including the topsoil, in vertical order would better and more rapidly restore hydrological properties with, in particular, slow infiltration and drainage of water and greater retention capacity; and (2) soil transfer treatments without topsoil, particularly those only involving altered bedrock, would lead to a marked difference from the reference steppe surrounding the restored site. The treatment involving soil surface topsoil transfer was expected to lead to finer grain size and greater biological activity and root density, close to that of the topsoil of the surrounding undisturbed steppe.

2. Materials and methods

2.1. Study area

Our study site is situated in the French Mediterranean region in pseudo-steppe vegetation, *Asphodeletum-fistulosii* phytosociological community (Molinier and Tallon, 1950), located in the plain of La Crau (Fig. 1). In the past, this pseudo-steppe covered around 50,000 ha (Molliex et al. 2013). Today, agricultural, military or industrial activities have reduced it to only a fragmented 10,500 ha, which served as the historical reference or target ecosystem (Clewelly and Aronson, 2013; Gann et al., 2018) as well as the donor site for soil transfers during earlier restoration projects (Alignan et al., 2018b; Alignan et al., 2018a; Bulot et al., 2017, 2014b; Coiffait-Gombault et al., 2012, 2011; Jaunatre et al., 2014).

The geographic area of the plain of La Crau corresponds to the former outlet of the Durance River generated by Quaternary deposits during the Pleistocene (Molliex et al. 2013). In this ecosystem, the combined effects of original soil conditions (see below for details), a dry and windy Mediterranean climate namely a warm temperate climate with dry and hot summer according to the Köppen-Geiger climate classification (Kottek et al. 2006) and many centuries of itinerant sheep grazing have shaped a unique Mediterranean dry grassland locally called "steppe"

(Devaux et al., 1983; Henry et al., 2010; Saatkamp et al., 2010). This steppe vegetation is dominated by perennial species and contains many annual species where trees cannot grow (Buisson and Dutoit, 2006). The dominant soil type has been identified as Mediterranean red soil (Baize et al. 2009), also referred as Haplic Cambisol (Calcaric) with a sandy-loam texture according to the World Reference Base for Soil Resources (WRB, 2015). covered more than 50 % by siliceous pebbles washed from the Alps. The topography is extremely flat and the original soil surface layer (around 20 cm of silt) was removed by wind erosion during the Late Glacial Period (Molliex et al., 2013). Three main pedological layers were previously identified under typical steppe vegetation (Bouteyre and Duclos, 1994) : 1. an organo-mineral topsoil (0–20 cm deep) largely composed of rounded siliceous pebbles (40–70 % of the surface) with a mixture of organic (2.5–3.5 %) and mineral matter; 2. a mineral horizon (20–40 cm deep) consisting of brown to red-brown soil that is very stony and differing from the bedrock in its greater alteration (presence of free Fe₂O₃) and from the surface topsoil in its structure; 3. altered bedrock (more than 40 cm deep) formed by the dissolution of the matrix hardened by rainwater rendered acidic by moving within the upper horizons. This bedrock, located 40–60 cm deep, is a stony conglomerate composed of pebbles contained in a calcium carbonate matrix (Molliex et al., 2013). It is between 5 and 35 m thick depending on location and prevents the plant root system from reaching the water table. The upper part of this conglomerate is an encrusting layer between 1 and 5 m thick resulting from the leaching of calcium by gravitational waters and/or from calcareous deposits from rising carbonate-rich groundwater (Bouteyre and Duclos, 1994).

2.2. Study site

The study site covered 5.5 ha (43°31'36.77"N, 4°53'04.50"E) where the grassland soil and vegetation were polluted by an oil spill in summer 2009 (Fig. 1). Restoration consisted in excavating the polluted soil down to the bedrock (72,000 tons) and taking it to a specialised treatment centre. In spring 2011, the excavated area was filled in by direct soil transfer (no stockpile) from intact dry grassland at a nearby active quarry (soil donor site) 4 km away (Fig. 1). The soil transfer was performed during a rainy period at a ratio of 1:1. The three main soil layers were reconstituted with their different degrees of profile complexity (reconstituted technosol), as derived from the original pedogenesis in the reference undisturbed steppe soil. This process enabled us to evaluate three treatments on plots of 100 m² randomly distributed over the 5.5 ha site and involving partial or total reconstitution of the soil original vertical organisation (for more details of this restoration process, see Bulot et al. 2014b and 2017), as follows:

- (1) "topsoil" (TS) comprising altered bedrock (35–40 cm deep) + subsoil (20–35 cm deep) + topsoil (0–20 cm deep);
- (2) "subsoil" (SS) comprising altered bedrock (20–40 cm deep) + subsoil (0–20 cm deep);
- (3) "altered bedrock" (AB) using only altered bedrock (from surface to 40 cm deep).

These treatments were also systematically compared to a positive reference: the surrounding undisturbed Mediterranean steppe (Step). The AB treatment was considered as a negative control, representing a situation with no ecological restoration requirements, as is typical in areas without environmental protection status.

After three years, the plant community recovery was more encouraging in TS and SS treatments in terms of composition, similarity and species richness but it was always better with the transfer of topsoil even if the vegetation structure were still different (Bulot et al. 2014b). In 2021, after ten years, these results were confirmed (Unpublished data) with a higher species-richness in characteristic plant species of the plain of La Crau on the study site but which were absent in the analyzed vegetation plots in the surrounding reference steppe.

2.3. Description of soil profiles

In spring 2016, a description of each soil profile was undertaken to analyse soil development six years after the soil transfer as compared to the undisturbed steppe. At the centre of one randomly selected plot for each treatment, a pedological pit was described after excavation down to the geological bedrock with a manual digger (approximately 1 m², 40 cm deep). The physical and biological states of plots were described according to the standard procedures using the STIPA 2000 data sheets of the *Institut National de la Recherche Agronomique* (Falipou and Legros, 2002) recommended by the *Association Française d'Etudes des Sols* (Baize and Jabiol, 2011). This was followed by designation of the soil profile and the pedological layers according to the World Reference Base for Soil Resources (Anjos et al., 2016).

2.4. Soil analyses

2.4.1. Soil sampling

Soil samples were taken and analysed immediately after the soil transfer in spring 2011, three years later in autumn 2013 and six years later in spring 2016 to assess changes in pH, nutrients (CaO, K₂O and P₂O₅) and particle sizes (silts, sands, clays). Five 200 g soil samples were extracted with a shovel (the soil was too hard and pebbly to use a soil core) to a depth of 10 cm in the five 100 m² plots for each treatment in order to make correlations between soil physico-chemical parameter and soil surface vegetation. To account for heterogeneity within plots, samples were homogenised by mixing three sub-samples taken randomly in each plot.

2.4.2. Physico-chemical parameters.

After air drying and sieving (2 mm) the soil, physico-chemical properties (pH water, CaO and K₂O and available phosphorous) were measured using standard methods: pH in soil; water ratio of 1:2.5; CaO and K₂O by the Metson method; and available phosphorous by the Olsen method (Metson, 1957; Olsen, 1954). Percentages of clay (< 2 µm), silt (2 – 50 µm) and sand (50 – 2000 µm) were determined according to the Robinson method (Baize, 2018) without decarbonation. Organic C content and total N content were quantified from bulk soil (before fractionation) in 2016 by dry combustion using a CHN elemental analyzer in the soil analysis laboratory of Arras (ISO 10694).

2.5. Hydrodynamic parameters

In spring 2016, hydrodynamic parameters were measured on restored (TS) or partially reconstituted soils (SS, AB) and compared to those of the surrounding steppe natural soil (Step). Three parameters were measured: (i) water retention curve, (ii) hydrophobicity, (iii) hydraulic conductivity near saturation.

2.6. Water retention curve

Because of the soil's high pebble content and low cohesion due to its sandy texture, it was not possible to use undisturbed soil cores or clods. Therefore, the water retention curve was measured on disturbed soil samples sieved to 2 mm using the usual pressure plate extractor method (Dane and Hopmans, 2002).

Nine samples were collected during the creation of pits for each of the four treatments:

- 2 samples in the Step pit, with only two horizons (TS and SS);
- 3 samples in the TS pit, one for each of the three horizons (TS, SS and AB);
- 2 samples in the SS pit (SS and AB);
- 2 samples in the AB pit: 1 in the upper layer between 0 and 10 cm, and 1 in the lower layer between 10 and 40 cm.

After soil drying and sieving to obtain homogeneous samples without gravel and pebbles, 5 to 8 g of soil were deposited on porous plates (3

replicates) in a ring of about 2.5 cm in diameter, rewetted up to full saturation and then placed in the pressure plate extractor. Five levels of pressure (0.01, 0.03, 0.1, 0.4 and 1.5 MPa) were applied by injecting nitrogen into the pressure chamber, causing the soil samples to drain. Once the outflow of soil water from the pressure chamber ceased, the water potential of the soil sample could be considered the opposite of the applied pressure. Pressure was then reduced to atmospheric, moist soil samples were removed from the porous plate and their gravimetric water content was calculated.

A continuous water retention curve using the van-Genuchten equation (van Genuchten, 1980) was then fitted to the experimental points. The van Genuchten equation for the water retention curve is:

$$\frac{W - W_r}{W_s - W_r} = \frac{1}{(1 + |h/h_g|^n)^m}$$

where W (kg kg^{-1}) is gravimetric soil water content, W_r (resp. W_s , kg kg^{-1}) is residual soil water content (resp. soil water content at saturation), h (Pa) is soil water potential, h_g (Pa) is a scale factor in soil porosity related to the air entry point, n (-) is a shape factor related to the size distribution of soil pores and $m = 1 - 1/n$.

2.7. Hydraulic conductivity

Infiltrometry tests were carried out in situ using a tension disk infiltrometer DECAGON® (see Angulo-Jaramillo et al., 2000, for a review on disc infiltrometers). We used the mini disk infiltrometer from Meter Environment Group equipped with a ring 8 cm in diameter to measure the near-saturated hydraulic conductivity. The infiltration tests were conducted applying 3 cm suction, which ensures the saturation of all soil pores with a diameter of less than 1 mm according to Jurin's law. As underlined by Jarvis et al., 2002, it is commonly observed that hydraulic conductivity increases by 1 to 3 orders of magnitude across a small pressure variation near saturation (10 cm to zero suction) due to rapid gravity-dominated flow in structural and biological macropores. However, following (Mohanty et al., 1997; Jarvis et al., 2002 or Vereecken et al., 2010), near-saturated hydraulic conductivity instead of the hydraulic conductivity at full saturation should be used as a scale factor of the unsaturated hydraulic conductivity curve to avoid large overestimates of the hydraulic conductivity of unsaturated soils. Mohanty et al. (1997) recommended a suction of 3 cm as a threshold between gravity-dominated flow (suction between 3 cm and 0 cm corresponding to near-saturated soils) and capillary-dominated flow (suction above 3 cm corresponding to drier soils). This threshold was therefore used in this study.

The experimental data were then analysed by BEST (Beerkan Estimation of Soil Transfer parameters) (Haverkamp et al., 1996) using the BEST-slope algorithm (Lassabatère et al., 2006).

These infiltrometry tests were performed in the field for each of the 9 soil layers identified above. Four replicas were performed for each horizon, around each of the 4 pits.

2.8. Hydrophobicity

We chose simpler methods, characterising soil wettability by calculating two different hydrophobicity indicators: Water Drop Penetration Time (WDPT), which is a measure of the persistence of soil hydrophobicity, and Molarity of Ethanol Droplet (MED), which measures the degree or intensity of soil hydrophobicity (Deurer et al., 2011). For both tests, soil samples were prepared as when measuring the water retention curve.

In the WDPT test, the time required for a drop of water to infiltrate the soil was measured using distilled water drops with a constant volume of 50 μL . Sieved dry soil samples were deposited in a ring of about 2.5 cm in diameter, with the soil levelled but not compacted. Three replicates of soil samples were used to characterise soil hydrophobicity

based on an infiltration time classification: hydrophilic (< 5 s), slightly hydrophobic (5 to 50 s), strongly hydrophobic (60 to 600 s), severely hydrophobic (600 to 3600 s) and extremely hydrophobic (> 3600 s) (Bisdorf et al., 1993).

In the MED test (King, 1981; Watson and Letey, 1970), several water-based solutions containing different volume percents of ethanol (from 0 to 36 %) were used to determine the infiltration rate of a drop of 50 μL deposited on the surface of the ground. The lower the concentration of ethanol in a solution infiltrating the soil in 3 s or less, the lower the soil's degree of hydrophobicity (Crockford et al. 1991). The solution containing the lowest volume percent of ethanol and infiltrating in 3 s or less reveals the degree of hydrophobicity of the soil (Crockford et al., 1991). The tests were therefore repeated, starting with higher volume percents of ethanol that were reduced until the drops seeped into the soil in 3 s or less. Degree of hydrophobicity of the soil was classified as follows (ethanol percentage indicated into brackets): 7 = extremely hydrophobic (36 %), 6 = very strongly hydrophobic (24 %), 5 = strongly hydrophobic (13 %), 4 = moderately hydrophobic (8.5 %), 3 = slightly hydrophobic (5 %), 2 = hydrophilic (3 %) and 1 = very hydrophilic (0 %) (Doerr, 1998).

2.9. Available soil water and water drainage

Two indicators were calculated to characterise soil water holding capacity: available soil water (AW), the maximum quantity of soil water available to plants, and water drainage (WD), the quantity of water that is quickly drained into the soil profile after rainfalls and is therefore only briefly available to plants (2 to 3 days).

AW under field conditions is the difference between the volumetric water content at field capacity θ_{cc} and the permanent wilting point θ_{pwp} , integrated over the full root depth. The permanent wilting point is usually estimated as the laboratory-measured volumetric water content corresponding to a soil water potential of -1.5 MPa. The field capacity was estimated here as the laboratory-measured volumetric water content at a soil water potential of -0.03 MPa, which is a typical value for sandy to medium-textured soils (Chessel et al., 2004; Dray and Dufour, 2007). Gravimetric water content at these two soil water potentials measured for the water retention curve were converted into volumetric water content using the measured bulk density via the cylinder method (4 replicates for each site and soil horizon). AW (mm) was calculated over full soil depth according to the different soil layers as follows: $AW = \sum_{\text{soillayers}_i} (\theta_{cc,i} - \theta_{pwp,i}) \cdot \Delta z_i$

where Δz_i is the thickness of soil layer i (mm).

The quantity of water drainage was calculated as the difference between volumetric water content at saturation θ_s and field capacity θ_{cc} , integrated over the full root depth. θ_s was calculated from the measured bulk density and the estimated W_s of the van Genuchten soil water retention curve equation. The same process used to calculate AW was applied to calculate WD (mm): $WD = \sum_{\text{soillayers}_i} (\theta_{s,i} - \theta_{cc,i}) \cdot \Delta z_i$

AW and WD were calculated for fine earth only, without accounting for the high pebble content in the soil profile. Computing the AW of the whole soil layer would require a precise estimation of the stone content of the soil layers on the different treatments. This estimation is difficult to obtain. Moreover, the stone content heterogeneity between treatments may be important and results only from variations during the initial soil transfer operation and not from different ecological processes that occurred since the initial soil transfer operation.

2.10. Statistical analyses

All data analyses were performed using the statistical software R version 3.6.1 (R Development Core Team, 2016) and the packages ade4 (Chessel et al., 2004; Dray and Dufour, 2007).

To characterise the physical and chemical parameters arising from the different soil treatments in 2016 and over three years (2011, 2013

Table 1

The 2016 soil profile standard descriptions (Jabiol and Baize, 2011) for each treatment and for the reference steppe: "Altered Bedrock" (AB), "SubSoil" (SS), "TopSoil" (TS), "Reference Steppe" (Step).

	Topsoil				Subsoil				Altered bedrock			
	AB	SS	TS	Step	AB	SS	TS	Step	AB	SS	TS	Step
Upper and lower limits (cm)	/	/	0–15	0–(15–20)	/	0–20	15–35	(15–20)–(40–45)	0–5	20–35	35–50	35–40
Texture	/	/	Silty	Silty	/	Silty	Silty	Silty	Sandy	Sandy	Sandy	Sandy
Sand size	/	/	0.05–0.10 mm	0.05–0.10 mm	/	0.05–0.10 mm	0.05–0.10 mm	0.05–0.10 mm	0.2–2 mm	0.2–2 mm	0.2–2 mm	0.2–2 mm
Coarse elements	/	/	60 %	70 %	/	60 %	60 %	80 %	90 %	90 %	95 %	90 %
			Rounded pebbles (2 to 6 cm)	Rounded pebbles (2 to 6 cm)		Rounded pebbles (2 to 6 cm)	Rounded pebbles (2 to 6 cm)	Rounded pebbles (2 to 6 cm)	Rounded pebbles (2 to 6 cm)	Rounded stones (6 to 20 cm)	Rounded pebbles (2 to 6 cm)	Rounded pebbles (2 to 6 cm)
			Unprocessed	Unprocessed		Slightly processed	Unprocessed	Unprocessed	Unprocessed	Slightly processed	Unprocessed	Unprocessed
			Any orientation	Horizontal orientation		Any orientation	Any orientation	Horizontal orientation	Any orientation	Any orientation	Any orientation	Horizontal orientation
			Reaction: acid	Reaction: acid		Reaction: acid	Reaction: acid	Reaction: acid	Reaction: acid and basic	Reaction: basic	Reaction: acid	Reaction: acid
Effervescence	/	/	no	no	/	Moderate, generalized	Weak, only in the calcareous matrix of pebbles	no	Extremely strong, in the calcareous matrix of pebbles, and in the soil skeleton	Very strong, in the calcareous matrix of pebbles, and in the soil skeleton	Strong, only in the calcareous matrix of pebbles	Strong, only in the calcareous matrix of pebbles
Structure	/	/	Subangular polyhedral (1–10 mm)	Subangular polyhedral (0.5–1.5 mm)	/	Subangular polyhedral (1–20 mm)	Subangular polyhedral (1–20 mm)	Angular polyhedral (0.5–1.5 mm)	Particulate	Particulate	Particulate	
Mechanical properties	/	/	Slightly brittle	Slightly brittle	/	Slightly brittle	Brittle	Slightly brittle	Very brittle	Slightly brittle	Not brittle	Brittle
			Hard	Slightly hard		Very slightly hard	hard	Slightly hard	Slightly hard	Slightly hard	Not hard	Very slightly hard
			Very compact	Loose		hard	Very compact	Loose	Loose	Slightly compact	Loose	Slightly compact
Soil moisture	/	/	Moist	Dry	/	Less moist	Dry	Dry	Fresh	Dry	Dry	Dry
Pore of aggregates	/	/	no	Many (50–200/dm ²) Very fine (d < 0.5 mm)	/	no	Not many (1–50/dm ²) Very fine (d < 0.5 mm)	Many (50–200/dm ²) Very fine (d < 0.5 mm)	Quite Numerous (50–200/dm ²) Very fine (d < 0.5 mm)	no	no	/
Global porosity	/	/	Slightly porous (2–5 %)	Porous (15–40 %)	/	Slightly porous (2–5 %)	Moderately porous (5–15 %)	Moderately porous (5–15 %)	Porous (15–40 %)	Porous (15–40 %)	Very porous (> 40 %)	/
Organic matters:	/	/	Medium abundance (1 à 4 %)	Medium abundance (1 à 4 %)	/	Medium abundance (1 à 4 %)	Low abundance (< 1 %)	Low abundance (< 1 %)	no	no	no	Undetermined abundance
			Very altered	Very altered		Very altered	Very altered	Very altered				Very altered
			Very fragmented	Invisible remains		Very fragmented	Invisible remains	Invisible remains				Invisible remains
Soil layer colour	/	/	Brown-yellow	Brown-red	/	Brown-red	Brown-red	Red-brown	Grey	Grey	Grey	Grey
Biological activities	/	/	no	A lot of	/	Fewer	no	Fewer	no	no	no	no
Plant roots	/	/	Few (8–16 / dm ²) Fine (0.5 to 2 mm)	Numerous (16–32/dm ²) Medium size (2 to 5 mm)	/	Few (8–16/dm ²) Mean size (2 to 5 mm)	Very few (< 8 dm ²) Fine (0.5 to 2 mm)	Few (8–16/dm ²) Fine size (0.5 to 2 mm)	Very few (< 8/dm ² , Very thin (< 0.5 mm), vertical,	Very few (< 8 dm ²) Fine (0.5 to 2 mm)	no	no
			Vertical	Vertical (2 to 5 mm)		Vertical	Vertical	Vertical	Inside the mass	Vertical		
			Normal form	Vertical		Normal form	Normal form	Normal form	Normal form	Normal form	Normal form	Normal form
			Healthy	Healthy		Healthy	Healthy	Healthy	Healthy	Healthy	Healthy	Healthy
			Inside aggregates	Healthy		Inside the mass	Inside aggregates	Inside aggregates	Constrained by physical obstacles	Inside aggregates	Constrained by physical obstacles	Constrained by physical obstacles
			Constrained by physical obstacles (pebbles)	Inside aggregates		Constrained by physical obstacles (pebbles)	Constrained by physical obstacles (pebbles)	Constrained by physical obstacles (pebbles)	(pebbles)	Constrained by physical obstacles (pebbles)	Constrained by physical obstacles (pebbles)	Constrained by physical obstacles (pebbles)
			No modification of the matrix	No modification of the matrix		No modification of the matrix	No modification of the matrix	No modification of the matrix	No modification of the matrix	No modification of the matrix	No modification of the matrix	No modification of the matrix
Earthworm galleries	/	/	no	no	/	no	no	Few vertical galleries (1–3/dm ²)	no	no	no	no
Limits between topsoil and subsoil	/	/	Regular	Corrugated	/	/	/	Corrugated	/	/	/	/
Limits between subsoil and altered bedrock	/	/	/	/	/	Very regular	Regular	Regular	/	/	/	/

Table 2

Physico-chemical properties (spring 2016) of four treatments realised in a restoration soil transfer experiment at a depth of 10 cm in the five 100 m² plots (La Crau plain, south-eastern France). The given values are means \pm standard errors; df, F, X², p correspond to the degree of freedom, the F value, the X² value and the p value resulting from ANOVAs tests or Kruskal-Wallis. Within a row, two boxes with a different letter have significantly different values according to Tukey post-hoc tests or Mann-Whitney-Wilcoxon tests.

	ANOVA/Kruskal-Wallis	Altered bedrock treatment (AB)	Subsoil treatment (SS)	Topsoil treatment (TS)	Topsoil of Reference steppe (Step)
Fine silts (g/kg)	F = 197.4; p < 0.001	90.8 \pm 4.76b	202.2 \pm 14.50a	198.6 \pm 4.83a	193.8 \pm 6.10a
Coarse silts (g/kg)	F = 119.2; p < 0.001	73.0 \pm 4.74d	121.4 \pm 6.73c	136.6 \pm 5.77b	155.6 \pm 10.40a
Clays (g/kg)	F = 284.0; p < 0.001	65.6 \pm 4.67c	189.2 \pm 14.89b	237.2 \pm 7.01a	204.0 \pm 10.17b
Fine sands (g/kg)	F = 204.5; p < 0.001	392.6 \pm 22.57a	204.2 \pm 1.79c	209.0 \pm 12.47c	257.0 \pm 9.35b
Coarse sands (g/kg)	F = 106.7; p < 0.001	378.0 \pm 14.37a	283.0 \pm 23.65b	218.6 \pm 17.43c	189.6 \pm 15.37c
Organic carbon (g/kg)	X ² = 16.37; p < 0.001	1.224 \pm 0.49c	9.898 \pm 0.68b	16.320 \pm 1.54a	17.160 \pm 2.02a
Total nitrogen (g/kg)	X ² = 16.58; p < 0.001	0.246 \pm 0.18c	1.032 \pm 0.05b	1.578 \pm 0.11a	1.516 \pm 0.13a
C:N (g/kg)	X ² = 17.6; p < 0.001	5.026 \pm 2.35b	9.594 \pm 0.30a	10.298 \pm 0.32a	11.340 \pm 0.35ab
Organic matter (g/kg)	X ² = 16.43; p < 0.001	2.122 \pm 0.85b	17.100 \pm 1.14ab	28.240 \pm 2.71a	29.700 \pm 3.48a
pH (g/kg)	X ² = 17.88; p < 0.001	8.770 \pm 0.04a	8.396 \pm 0.06ab	7.260 \pm 0.21b	6.844 \pm 0.16b
CaO (g/kg)	X ² = 16.91; p < 0.001	9.198 \pm 0.10a	6.766 \pm 2.08b	2.622 \pm 0.30c	2.124 \pm 0.12c
P ₂ O ₅ (g/kg)	F = 9.05; p < 0.001	0.005 \pm 0.00b	0.009 \pm 0.002a	0.011 \pm 0.002a	0.006 \pm 0.001ab
CEC Metson (g/kg)	F = 384; p < 0.001	2.306 \pm 0.09c	7.494 \pm 0.63a	9.046 \pm 0.26b	8.676 \pm 0.56b
K (g/kg)	X ² = 15.73; p < 0.01	0.030 \pm 0.00b	0.116 \pm 0.03a	0.190 \pm 0.03a	0.120 \pm 0.04a

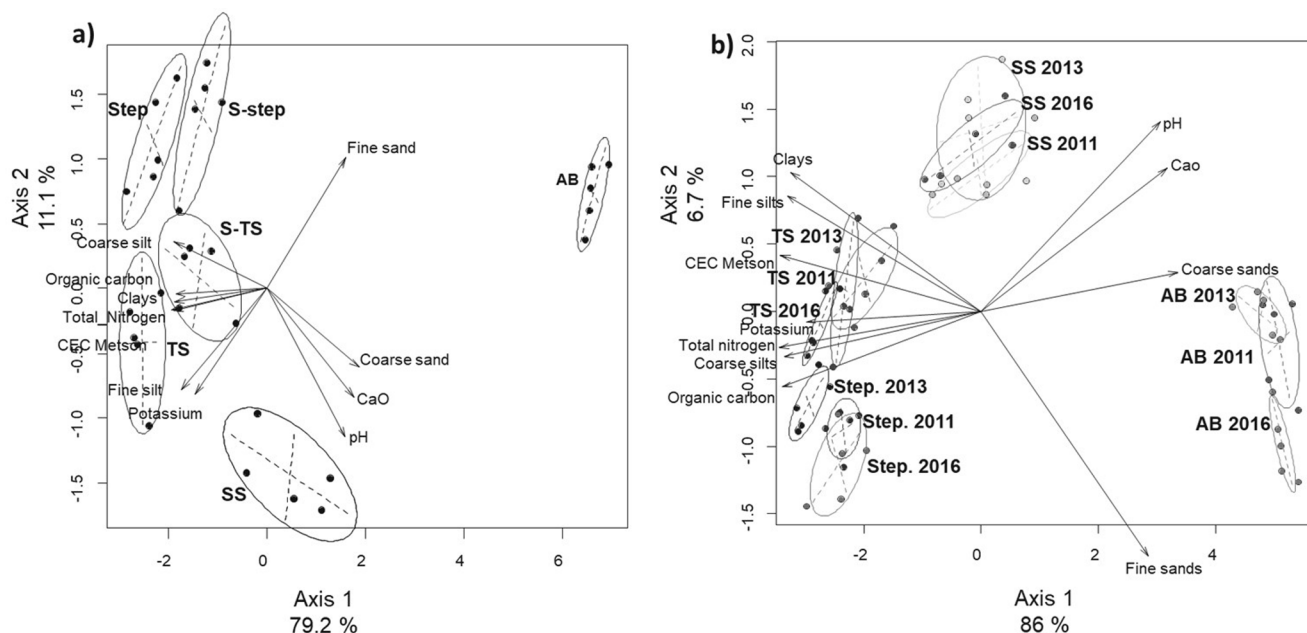


Fig. 2. PCA ordinations based on 5 soil samples grouped by treatment with ellipsis (n = 30) a) in 2016 only and b) for the 3 years of soil sampling (2011, 2013, 2016) (Step: topsoil of the reference steppe, S-Step: subsoil layer of the reference steppe (S-Step), TS: topsoil layer of the topsoil treatment, SS-TS: subsoil layer of the topsoil treatment (S-TS), SS: subsoil of the subsoil treatment, and the altered bedrock treatment (AB)).

and 2016), a multivariate Principal Component Analysis (PCA) was performed. Data on physical and chemical parameters and on hydraulic conductivity values were analysed via one-way analyses of variance (ANOVA tests). These were followed by post hoc Tukey's tests, performed to compare means ($\alpha = 0.05$) between the different soil treatments or soil layers in each soil treatment. When data did not conform to parametric conditions, means normality of model residuals (Shapiro's tests) and homogeneity of variance (Mauchly's tests), Kruskal-Wallis non-parametric tests were used, followed by pairwise Mann-Whitney-Wilcoxon comparisons with a p-value adjustment according to Benjamini-Hochberg (Benjamini and Hochberg, 1995).

Finally, Pearson's correlations tests were applied to the measured values of hydraulic conductivity, retention of water for the different suction strengths, particle sizes and organic matter contents.

3. Results

3.1. Soil profile descriptions

The soil profiles realised in 2016 on the different soil transfer treatments (AB, SS and TS) revealed strong differences from the reference steppe soil in composition and structure even six years after soil transfer (Table 1). When the original vertical order of soil layers was less faithfully reconstituted, the differences from the reference steppe became even more pronounced.

Principally, compared to the soil of the reference steppe, the horizons of the soil transfer treatments were very little colonised by plant roots, especially those of the dominant tussock perennial grass of the steppe (*Brachypodium retusum* L.), and they showed very few biological activities and earthworm galleries. Their porosity was also lower than in the steppe reference soil. In addition, the limits between the different soil layers remained more distinct in the soil transfer treatments, for

Table 3

Physico-chemical characteristics (Spring 2016) in the subsoil layer of four treatments realised in a restoration soil transfer experiment at a depth of 10 cm (SS treatment) and 20 cm (S-TS and S-Step treatments) in the La Crau plain, southeastern France. The given values are means \pm standard errors; df, F, X^2 , p correspond to the degree of freedom, the F value, the X^2 value and the p value resulting from ANOVAs tests or Kruskal-Wallis. Within a row, two boxes with a different letter have significantly different values according to Tukey post-hoc tests or Mann-Whitney-Wilcoxon tests.

	ANOVA/ Kruskal- Wallis	Subsoil treatment (SS)	Subsoil of Topsoil treatment (S-TS)	Subsoil of Reference steppe (S-Step)
Fine silts (g/kg)	F = 2.709; p > 0.05	202.2 \pm 14.50a	186.4 \pm 3.38a	188.2 \pm 11.56a
Coarse silts (g/kg)	$X^2 = 9.64$; p < 0.01	121.4 \pm 6.73b	142.2 \pm 6.91a	143.0 \pm 7.65a
Clays (g/kg)	F = 17.45; p < 0.01	189.2 \pm 14.89b	238.8 \pm 12.15a	237.6 \pm 17.85a
Fine sands (g/kg)	F = 12.97; p < 0.001	204.2 \pm 1.79b	203.4 \pm 17.73b	239.4 \pm 13.09a
Coarse sands (g/kg)	F = 28.99; p < 0.001	283.0 \pm 23.65a	229.2 \pm 9.73b	191.8 \pm 20.81c
Organic carbon (g/kg)	F = 14.62; p < 0.001	9.898 \pm 0.68b	12.160 \pm 0.38a	10.302 \pm 0.94b
Total nitrogen (g/kg)	F = 26.28; p < 0.001	1.032 \pm 0.05b	1.266 \pm 0.03a	1.054 \pm 0.08c
C:N (g/kg)	F = 0.58; p > 0.05	9.594 \pm 0.30a	9.620 \pm 0.19a	9.748 \pm 0.23a
Organic matter (g/kg)	F = 15.63; p < 0.001	17.100 \pm 1.14b	21.060 \pm 0.63a	17.820 \pm 1.60b
pH (g/kg)	F = 97.27; p < 0.001	8.396 \pm 0.06a	7.546 \pm 0.23b	6.860 \pm 0.19c
CaO (g/kg)	F = 18.78; p < 0.001	6.766 \pm 2.08a	2.984 \pm 0.52b	2.270 \pm 0.28c
P ₂ O ₅ (g/kg)	$X^2 = 14.0$; p < 0.001	0.009 \pm 0.002a	0.010 \pm 0.000a	0.000 \pm 0.000b
CEC Metson (g/kg)	F = 6.59; p < 0.001	7.494 \pm 0.63b	8.524 \pm 0.17a	8.738 \pm 0.76a
K (g/kg)	F = 10.34; p < 0.01	0.116 \pm 0.03ab	0.110 \pm 0.00a	0.078 \pm 0.01b

example in the transferred topsoil (TS) (Table 1).

3.2. Physico-chemical parameters

Six years after the soil transfer, the soil transfer treatments showed significant differences in soil physico-chemical parameters from the topsoil of the reference steppe (Table 2, Fig. 2a). The PCA ordination based on soil properties significantly discriminated between treatments. Axis 1 (79.2 % of inertia) separated treatments with a topsoil transfer (Step, TS) from the negative control (AB) without topsoil transfer. This separation can be explained partly by the significantly larger quantities of chemical elements (N, P, K) in the TS and Step topsoil treatments, as well as by a significantly finer grain size for these treatments (Table 2). Conversely, when the topsoil was missing, the substrates were characterised by significantly larger quantities of fine and coarse sands, significantly higher CaO content and higher pH (Table 2). Axis 2 (11.1 % of inertia) clearly separated the soils of the reference steppe from the TS and SS treatments for all the parameters measured.

Additionally, the physico-chemical characteristics of the subsoil of the topsoil treatment (S-TS) were closer of those of the topsoil of the reference steppe. The surface layer of the subsoil treatment (S-SS) contained a significantly greater quantity of coarse sands, with higher CaO content and pH, than the subsoil of the reference steppe (S-Step). This may be due to partial mixing of the two transferred soil layers (S and AB

layers) during soil transfer in the SS treatments. The subsoil grain size of the topsoil treatment (S-TS) was also finer than that of the similar non-disturbed soil layer in the reference steppe (Table 3, Fig. 2a). Similar results were obtained for 2011, 2013 and 2016 (Fig. 2b).

3.3. Water infiltration, hydraulic conductivity and hydrophobicity

Water infiltration was slower in the surface layers for the TS and SS treatments and for the reference steppe. It was faster in SS and AB soil layers (except for the AB soil layer of TS treatment), particularly in the AB treatment and the AB soil layer of the SS treatment. As a rule, water infiltration was slower in any layer of the reference steppe than in the corresponding layers of all soil transfer treatments (Figs. 3 & 4).

Near-saturated hydraulic conductivity values were checked against experimental infiltration data: low conductivity values correspond to slow cumulative infiltrations and vice-versa. For a given depth, hydraulic conductivity was therefore lower in the soil of the reference steppe than in the transferred soils. In addition, there was a significant difference between the surface layer and deeper layers of the AB treatment (Fig. 5). Positive correlations were also found between hydraulic conductivity and sand content, whereas there were negative correlations between hydraulic conductivity and higher clay, silt and organic matter content (Table 4).

MED and WDPT tests for hydrophobicity always yielded water infiltration times of under three seconds, whatever the treatment sampled, indicating very hydrophilic soils.

3.4. Bulk density and water retention

Mass water content variability was fairly low in soil layers of the same type (AB or TS and SS soil layers), except at saturation, where amplitudes were greater (Figs. 6 & 7, Table 5).

For a given treatment, the water content at a given suction decreased with depth, except for the AB treatment, where the same water content was observed in the surface layer and in the deeper layers.

The water content of any soil layer of the Step treatment exceeded that of the corresponding layer of the TS and AS treatments at suction of between 0 cm (saturation) and 1 bar. The difference was most pronounced at saturation, decreasing when suction was 0.1, 0.33 and 1 bar. At high suction (dry soil), there was no difference between soil layers, nor between Step, TS and SS treatments.

Whatever the treatment and the water suction, water content in the AB soil layers was always lower than in the TS and SS soil layers. Virtually no difference was observed for the AB treatment between the measurements obtained at the surface and 10 cm deep, whatever the soil suction: the two water retention curves are very similar (see Fig. 7.c).

Parameters fitted to the water retention curves are presented in Table 5. Residual water content was always set to 0 g g⁻¹ in the BEST algorithm, consistent with the sandy or coarse silty texture of the soil. The measurements presented in Fig. 6 show lower water content at saturation for the AB soil layer than for the other layers, whatever the treatment: W_s ranged between 0.320 and 0.402 g g⁻¹ for the AB layer, whereas it ranged between 0.447 and 0.494 g g⁻¹ for TS and SS layers. The n parameter was almost constant, ranging between 1.211 and 1.290 whatever the treatment, typical values for medium fine/silty soil textures (Wösten et al., 2001). The hg parameter depended on air entry point: the higher the hg, the smaller the soil pore size at the air entry point.

With regard to the retention curves, water still seemed better retained in the soil of the reference steppe than in that of the TS treatment (except in the AB layer, where retention capacity was lower). Water retention was lower in the SS and AB treatments (Figs. 6 & 7, Table 5).

Negative correlations were measured between water retention and sand content, and positive correlations between water retention and clay, silt and organic matter contents (Table 4).

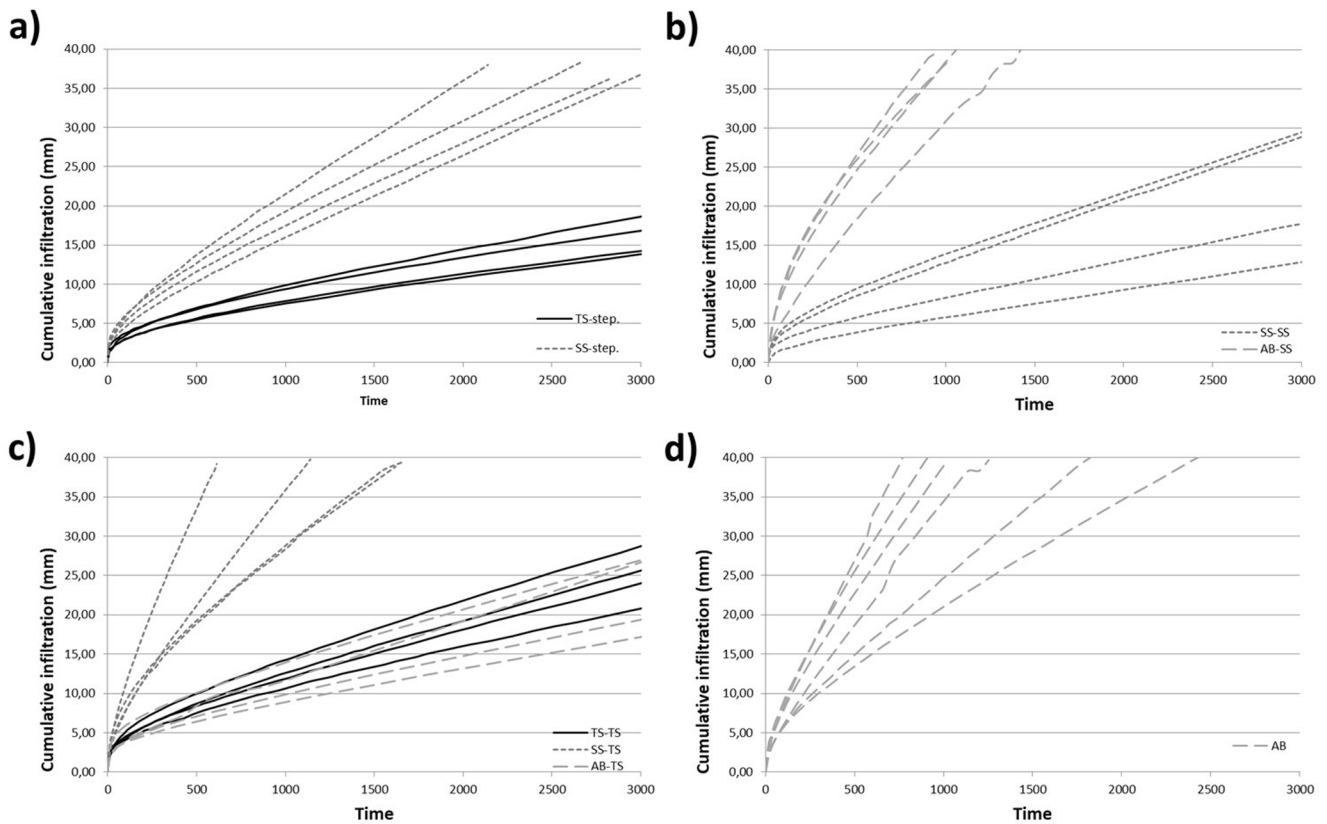


Fig. 3. Experimental data on cumulative infiltration (in mm) as a function of time (in seconds) by treatment: a) reference steppe (Step), b) topsoil treatment (TS), c) subsoil treatment (SS), and d) altered bedrock treatment (AB). Solid lines stand for the surface layer (topsoil), thin dotted lines for the mineral layer (subsoil) and spaced dotted lines for the altered bedrock layer.

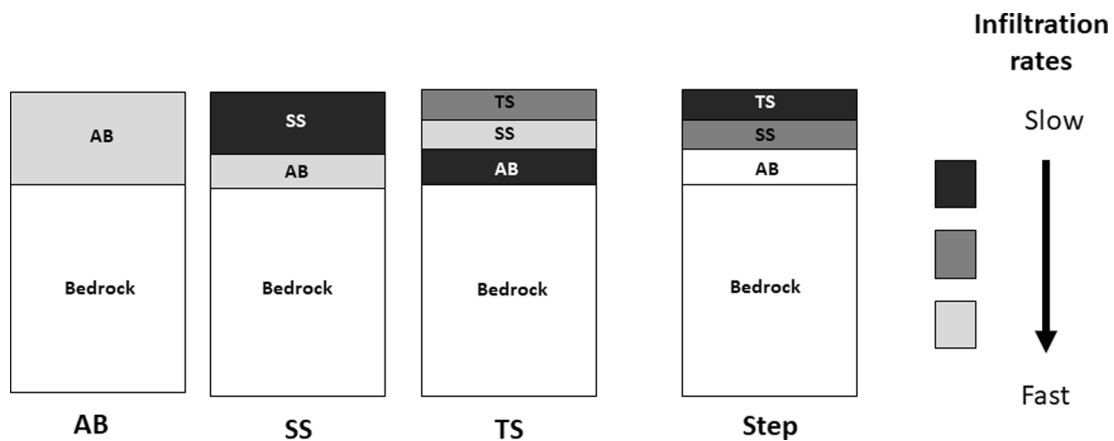


Fig. 4. Scheme illustrating the infiltration rates in the different soil layers tested in situ (TS: topsoil, SS: Subsoil, AB: altered bedrock) of each treatment (Step: reference steppe, TS: topsoil treatment, SS: subsoil treatment, AB: altered bedrock treatment). Lighter grey indicates a slow infiltration rate.

3.5. Available water and water drainage

The quantity of soil water available to the plant (AW) was higher in the soil of the reference steppe and lower in the AB treatment. The TS treatment was closer to Step., although AW was also lower (Table 6a).

There was less difference between treatments in quantity of water drainage (WD), although values were higher for Step and TS and lower for AB. As with hydraulic conductivity, there was a distinction between the AB soil layer of the TS treatment and the AB soil layers of the SS and AB treatments (Table 6b).

4. Discussion

Soil profiles reveal strong differences in composition and structure between the transferred soils and the reference steppe soil that remain even six years after soil transfer. This still confirms findings in 2013, three years after the soil transfer (Bulot et al., 2017). The pedological layers of all soil transfer treatments remain poorly colonised by the roots of the tussock perennial grass (*B. retusum*), despite its clear dominance even in the deepest pedological layers of the reference steppe. Soil analyses show that, in the soil layers of the reference steppe and in the transferred topsoil, where surface vegetation cover is greatest (Bulot et al., 2014b), organic matter content is significantly higher and visible,

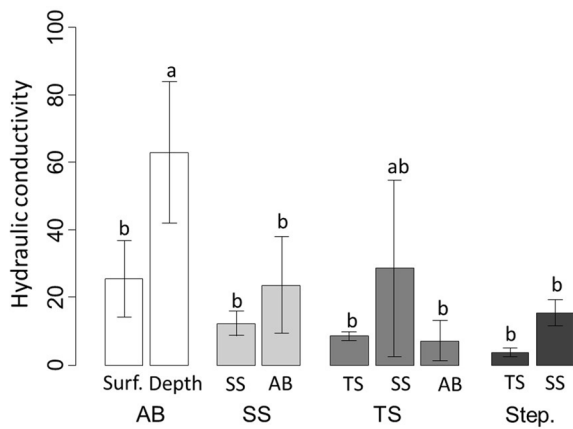


Fig. 5. Near-saturated hydraulic conductivity (mm h⁻¹) values obtained with the BEST-slope model by soil layer (TS: topsoil, S: subsoil, Surf: 0–10 cm deep, depth: 10–40 cm deep) and by treatment in 2016 (Step: reference steppe, TS: topsoil treatment, SS: subsoil treatment, AB: altered bedrock treatment). The measured values are means \pm standard errors; degree of freedom = 8, F value = 5.655 and p-value < 0.001 resulting from an ANOVA test. Two bars with a different letter have significantly different values according to Tukey post-hoc tests.

in line with the soil profile description of 2016. In contrast, where only altered bedrock was transferred, vegetation cover remains very low (Bulot et al., 2014b), as does soil organic matter content and biological activity (no traces of roots or earthworm galleries, etc.).

The differences in root colonisation observed between the soil treatments and the reference steppe could partially be due to the lower vegetation cover, as well as to soil compaction by construction machinery in spring 2011 during soil layer spreading. Soil compaction has been shown to inhibit the growth and distribution of plant roots, thereby modifying the dynamics of hydrodynamic properties (Alaoui et al., 2011; Alaoui and Helbling, 2006; Berli et al., 2008; Etana and Håkansson, 1994; Hamza and Anderson, 2005; Lipiec et al., 2009; Unger and Kaspar, 1994). Biological activity also appears extremely low in all the transferred soil layers, with very few earthworm galleries or turricules in comparison with the soil layers of the reference steppe. This too can be attributed to the impacts of the soil transfer in 2011, which destroyed all the *B. retusus* plants (Bulot et al., 2014b) and most of the dominant anecic earthworms, *Hormogaster elisae* (Oligochaeta, Hormogastridae), present in the soil of the reference steppe (Dutoit, pers. obs). In 2016, no wormcasts were observed at soil surface throughout the restored area (Bulot, pers. obs.). The absence of earthworm activity due to soil compaction could also explain some of the differences in water vertical circulation in the different soil layers (Capowiez et al., 2021; Edwards, 2002; Ernst et al., 2009; Hallam and Hodson, 2020; Radford et al., 2001). Certain invertebrates, through their ability to move in the soil and to build biogenic structures, can induce changes in the soil's organic and mineral nature (Bottinelli et al., 2015; Jouquet et al., 2006; Lee and Foster, 1991; Wills and Landis, 2018), as already proved for ants like *Messor barbarus* L. (Hymenoptera, Formicidae) (Azcarate and Peco, 2007; Martín-Perea et al., 2019) reintroduced in the restored area (Bulot

et al., 2014a, 2016; De Almeida et al., 2020). The soil profiles show that porosity remains lower in the pedological layers of the transferred soil than in the different layers of the undisturbed surrounding reference steppe soil. The limits between the different soil layers are also revealed as more distinct in the soil transfer treatments than in the soil of the reference steppe. Earthworms or ants and galleries from decomposed roots are known to contribute to the formation of soil macropores that can improve aeration and water infiltration (Capowiez et al., 2021; Colloff et al., 2010; De Almeida et al., 2020; Eldridge et al., 2010; Ernst et al., 2009; Lavelle, 1997; Lee and Foster, 1991; Lobry de Bruyn and Conacher, 1994).

Concerning physico-chemical soil parameters, compared to the reference steppe, the higher calcium and pH content in the SS and AB treatments could be explained by the solubilisation of calcium carbonates from the calcareous matrix of the natural conglomerate geological substratum or from calcareous stones degraded during transfer. Indeed, the subsoil and the altered bedrock were originally more similar to the calcareous conglomerate. Similar results were previously obtained in the same area in formerly cultivated fields (Jaunatre et al., 2014) and after trenching for pipelines (Coiffait-Gombault et al., 2012). The increase in organic C content from the negative control ("altered bedrock") to the "topsoil" treatment confirms our assumption that the deeper soil layer would contain less organic matter, and therefore less organic carbon (Baize et al., 2009). For these same parameters, the significant differences between the subsoil of the topsoil treatment and the subsoil of the reference steppe could also be explained by the mixing of topsoil and subsoil during lorry transport. All these findings echo those obtained in 2013 (Bulot et al., 2017) and show no significant difference from the situation prevailing in 2016.

Similarly, differences in soil hydrodynamic properties were found. Water infiltrates more slowly into the reference steppe soil, and for a given depth, conductivity is also lower than in the transferred soils. Reference steppe water retention capacity is greater, with in particular a larger quantity of available soil water. The quantity of water drainable is also higher there, especially for the transferred topsoil layer.

The TS treatment was the most faithful possible reconstitution of the reference steppe soil, with the same vertical organisation of the three pedological layers. However, the infiltration rate is observed to be faster overall than in the reference steppe, especially in the sub-soil layer, where the rates are close to those measured in the altered bedrock layers of the SS and AB treatments. TS drainage rates are similar to those of the reference steppe, but water retention capacity is better in the transferred topsoil layer than lower down (transferred S and AB layers), since the greater quantity of available soil water means that water infiltrates the deeper layers quickly. In contrast, the reconstitution in the SS and AB treatments was less faithful to the reference steppe soil. As a consequence, drainage is rather rapid and retention capacities lower. In the SS treatment, differences from the soil of the reference steppe mostly concern the altered bedrock layer. SS treatment properties show differences between the surface and the deeper layers that might indicate temporal evolution, with the forming of a vertical stratification of the soil.

Water retention results show little difference between treatments but more between the different vertical soil layers. In the same way, water infiltration and water drainage rates are faster for subsoil and altered

Table 4

Coefficients of correlations (Pearson method) between measured values for near-saturated hydraulic conductivity (Cond), retention of water for the different suctions (Ret_Suc0, Ret_Suc01, Ret_Suc033, Ret_Suc1, Ret_Suc4, Ret_Suc15), particle sizes and organic matter content in 2016. All correlations are significant ($p < 0.05$).

	Cond	Ret_suc0	Ret_suc01	Ret_suc033	Ret_suc1	Ret_suc4	Ret_suc15
Clays	-0.75	0.93	0.89	0.85	0.89	0.96	0.94
Fine silts	-0.73	0.91	0.92	0.91	0.94	0.99	0.99
Coarse silts	-0.86	0.92	0.95	0.92	0.94	0.92	0.90
Fine sands	0.65	-0.89	-0.85	-0.85	-0.89	-0.97	-0.99
Coarse sands	0.84	-0.89	-0.91	-0.86	-0.88	-0.87	-0.83
Organic matter	-0.85	0.91	0.92	0.87	0.89	0.89	0.85

Soil Water Retention Curve (sieved soil < 2mm) - ECCOREV - layers properties

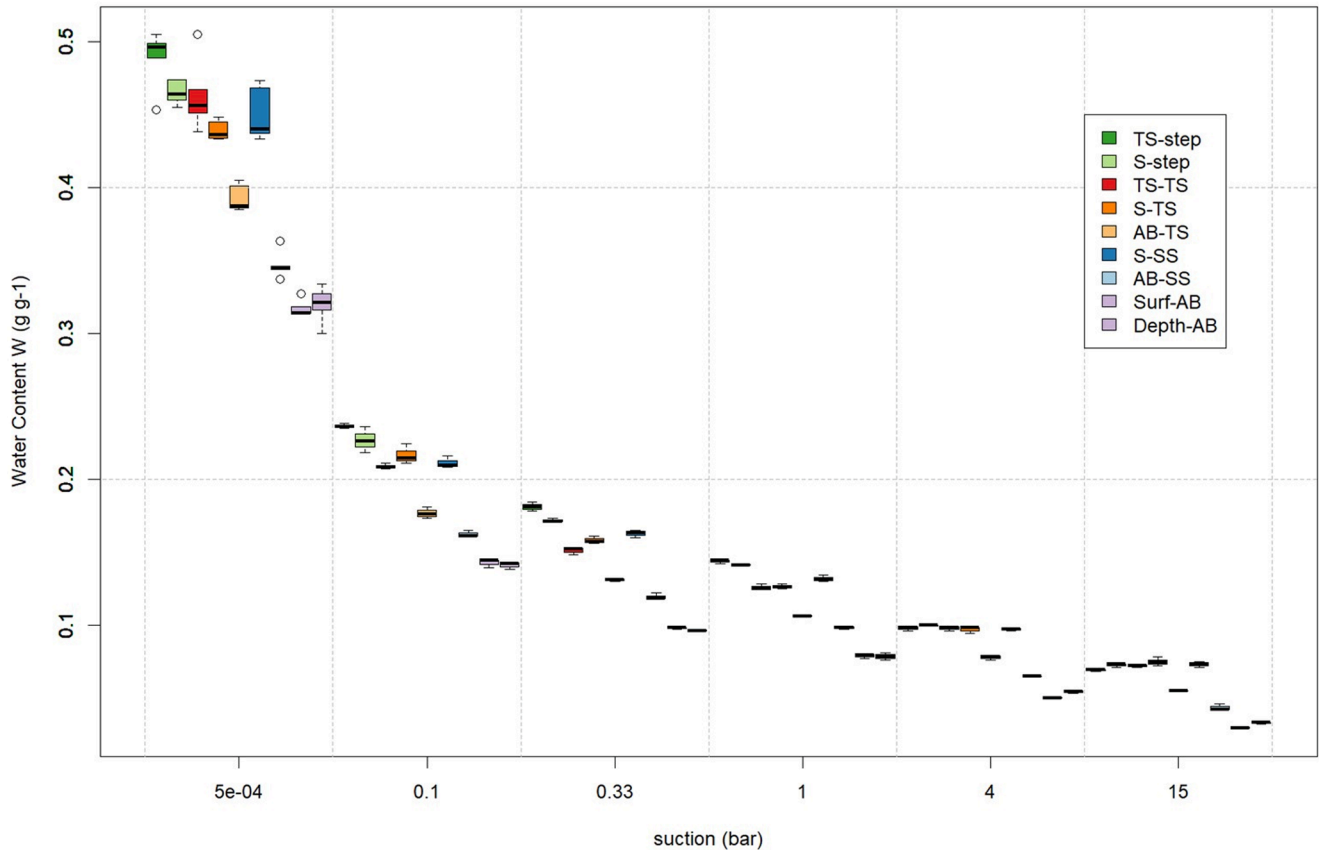


Fig. 6. Retention measures by soil layer (TS: topsoil, S: subsoil, AB: altered bedrock) and by treatment: a) Topsoil in topsoil treatment (TS-TS or in reference steppe (TS-Step); b) Subsoil in subsoil treatment (SS-SS), topsoil treatment (SS-TS) and reference steppe (SS-Step); and c) Altered bedrock at surface or deeper in the altered bedrock treatment (AB 0–10 cm and AB 10–40 cm), the subsoil treatment (AB-SS) and topsoil treatment (AB-TS). They represent the mass water content w (g g^{-1}) for a given suction strength (bar).

bedrock layers and for the altered bedrock layer of the SS treatment. This could be attributed to differences in soil texture and organic matter content (see above). Along with bulk density and organic matter content, soil texture is the characteristic most commonly used to predict soil water retention and hydraulic conductivity: hydraulic conductivity decreases and water retention increases with an increased fraction of fine grains (McBride and Mackintosh, 1984; Vereecken, 1995; Wösten et al., 2001). In fact, soils with a sandy texture retain little water and become desaturated more quickly. However, this is not the case for the altered bedrock layer of the AB treatment, where there is slow infiltration. Two reasons for these differences can be suggested, particularly when making comparisons with the altered bedrock layer of the SS treatment. First, discrepancies in the implementation of soil transfer may have generated greater compaction in some treatments. Second, a supply of finer materials from the transferred topsoil or subsoil layers may have reduced the porosity of the altered bedrock layer, thus reducing the rate of infiltration.

In addition, for all treatments, the measures of available soil water and water drainage only considered the fine fraction of the soil. However, these types of soils normally contain a high concentration of pebbles not conducive to water retention. The real values are then certainly lower.

These changes in soil hydrodynamic properties of soils with respect to intact soils can potentially affect the regeneration of the vegetation being restored. Several studies have already shown that water availability can not only limit the production of plant biomass and productivity, but also modify the species composition and the species richness

of vegetation (Deng et al., 2016; Enright et al., 2005; McKenzie et al., 2004; Pueyo et al., 2013; Thompson et al., 2010; Wu et al., 2014). The slower water drainage in the intact soil no doubt afforded the plant species more time to extract enough water for growth, whether it was easy or difficult to access. In addition, the drainable water may have remained accessible to vegetation in some locations, depending on the infiltrability of the bedrock, which varies greatly. In the TS treatment, the most similar to the reference steppe, changes in hydrodynamic properties may have slowed down regeneration of the plant community: according to the soil profile, six years after soil transfer, the roots of plants such as the dominant tussock perennial grass *B. retusum* have not yet reached the subsoil layer.

In the subsoil treatment, and even more so in the altered bedrock treatment, faster water infiltration and lower water retention capacity could act as a brake on vegetation development. These observations are consistent with the recovery of plant communities monitored previously on the study site (Bulot et al., 2014b). In turn, vegetation may modify the soil's hydrodynamic properties (Fischer et al., 2014, 2015; Gould et al., 2016). Vegetation has a strong influence on the infiltration capacity of soils, mainly through plant roots, and the presence of vegetation has been shown to increase soil infiltrability (Archer et al., 2002, 2016; Liu et al., 2019a; Meek et al., 1992). The limited colonisation by plant roots observed in the deeper layers of the transferred soil profiles may therefore also explain their hydrological functioning differences from the reference steppe.

In addition, adequate vegetation cover can enhance the regeneration of hydrodynamic properties. Vegetation can physically protect the soil

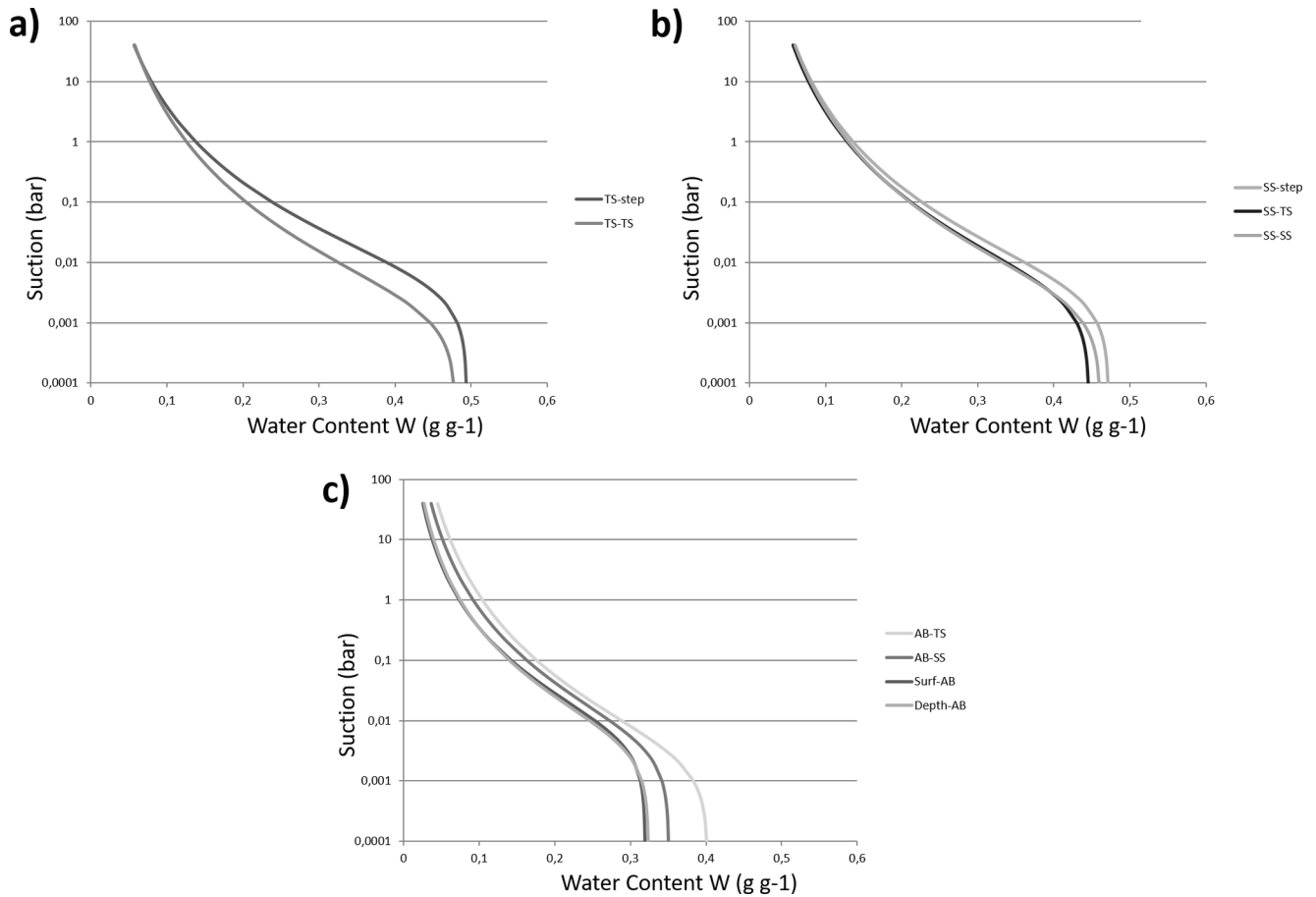


Fig. 7. Retention curves by soil layer (TS: topsoil, S: subsoil, AB: altered bedrock) and by treatment: a) Topsoil in topsoil treatment (TS-TS) or in reference steppe (TS-Step); b) Subsoil in subsoil treatment (SS-SS), topsoil treatment (SS-TS) and reference steppe (SS-Step); and c) Altered bedrock at surface or deeper in the altered bedrock treatment (AB 0–10 cm and AB 10–40 cm), the subsoil treatment (AB-SS) and topsoil treatment (AB-TS). They represent the mass water content w (g g^{-1}) for a given suction strength (bar).

Table 5

Parameters of the retention curves obtained using the Van-Genuchten model and the bulk density (kg dm^{-3}) measured in the three main soil layers (topsoil, subsoil and altered bedrock) for the four tested treatments: transferred topsoil (TS), subsoil (SS) and altered bedrock (AB) treatments and the soil of the reference steppe (Step). W_r is always set to 0 g g^{-1} due to the sandy/silty texture of the fine soil.

Soil layer Treatment	Topsoil				Subsoil				Altered bedrock			
	AB	SS	TS	Step	AB	SS	TS	Step	AB	SS	TS	Step
W_r (g g^{-1})			0	0	0	0	0	0	0	0	0	
W_s (g g^{-1})			0.480	0.494	0.461	0.447	0.472		Surf: 0.320 Depth: 0.324	0.351	0.402	
hg (bar)			0.002	0.005	0.002	0.003	0.004		Surf: 0.006 Depth: 0.005	0.005	0.003	
n (-)			1.211	1.239	1.211	1.219	1.222		Surf: 1.290 Depth: 1.273	1.250	1.228	
$m = 1-1/n$			0.174	0.193	0.174	0.179	0.182		Surf: 0.225 Depth: 0.224	0.200	0.185	
Measured bulk density (kg dm^{-3}), SD in brackets (n = 4 for SS, TS and Step; n = 2 for AB)			1.53 (0.06)	1.40 (0.06)	1.39 (0.15)	1.40 (0.07)	1.41 (0.06)		Surf: 1.57 (0.06) Depth: 1.59 (0.22)	1.70 (0.12)	1.65 (0.04)	

Table 6

Available water AW and water drainage WD measured a) in the different treatments (Step: reference steppe, TS: topsoil treatment, SS: subsoil treatment and AB: altered bedrock treatment) and b) in the different soil layers of each treatment (TS: topsoil, SS: subsoil, AB: altered bedrock) for an overall soil thickness of 40 cm.

a)				
	Step	TS	SS	AB
AW (mm)	59.3	52.5	51.9	42.9
WD (mm)	175.0	178.0	165.2	144.3
b)				
	Treatment	TS	SS	AB
AW (mm)	Step	6.08	5.78	
	TS	5.41	5.23	5.15
	SS		5.31	5.08
	AB			4.29
WD (mm)	Step	17.73	17.28	
	TS	19.76	15.98	17.69
	SS		17.39	15.65
	AB			14.43

surface against erosion (Loch, 2000; Puigdefábregas, 2005; Sanchez and Karl Wood, 1989) by limiting the impact of raindrops, incorporating runoff energy and fostering the formation and stabilisation of soil aggregates, for example (Bochet et al., 1999). Infiltration rates and water retention capacity can thus be increased, in particular through increased organic matter content in soils and enhancement of the nutrient cycle and the activity of microorganisms (Cerdà, 1997; Fischer et al., 2014; Liu et al., 2019b). Microbial activity is also particularly important in arid and semi-arid areas, where vegetation forms a mosaic with areas of bare soil (Cerdà, 1997; Maestre and Cortina, 2002), as already shown here for the steppe vegetation (Bulot et al., 2017; Martin et al., 2022).

5. Conclusion

Six years after soil transfer, hydrodynamic properties in these reconstituted soils, with or without topsoil replacement, remain different from those in the reference steppe. A difference in hydrological functioning is revealed: the reconstituted soils contain less available soil water and drain more quickly than the undisturbed soil. This can be explained in particular by differences in plant cover and lower organic matter content in the transferred soils or by reduced biological activity. The most encouraging results were obtained when the three main layers of soil were reconstituted, with a direct topsoil transfer from the donor site in spring. Nevertheless, soil transfer is not a sustainable restoration process, imposing a very heavy economic and environmental burden and requiring the destruction of the donor site.

While good results have been obtained previously on the regeneration of soil surface vegetation (Bulot et al., 2014b), such recovery could be compromised in the long term by soil hydrological dysfunction. This may increasingly become an issue under more frequent water stresses due to global warming.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgement

We thank the *Société du Pipeline Sud-Est Européen*, the *Conseil Général*

des Bouches-du-Rhône and the Région Provence-Alpes-Côtes-d'Azur (PACA), SFR Tersys and FR Eccorev for funding our work. We also thank the *Conservatoire des Espaces Naturels (PACA)*, the *Chambre d'Agriculture PACA* and the *Réserve Naturelle Nationale des Coussouls de Crau* for permission to carry out our study and for access to the study site. We are also grateful to all PhD students, student interns, technicians for assistance with the field work. Thanks to Marjorie Sweetko for correcting the English of previous versions of this paper.

References

- Adriaens, D., Honnay, O., Hermy, M., 2006. No evidence of a plant extinction debt in highly fragmented calcareous grasslands in Belgium. *Biol. Conserv.* 133, 212–224. <https://doi.org/10.1016/j.biocon.2006.06.006>.
- Alaoui, A., Helbling, A., 2006. Evaluation of soil compaction using hydrodynamic water content variation: Comparison between compacted and non-compacted soil. *Geoderma* 134, 97–108. <https://doi.org/10.1016/j.geoderma.2005.08.016>.
- Alaoui, A., Lipiec, J., Gerke, H.H., 2011. A review of the changes in the soil pore system due to soil deformation: A hydrodynamic perspective. *Soil Tillage Res.* 115–116, 1–15. <https://doi.org/10.1016/j.still.2011.06.002>.
- Alard, D., Poudevigne, I., 2002. Biodiversity in changing landscapes: From species to patch assemblages to system organization. *Application of geographic information systems and remote sensing in river studies* 9–24.
- Alday, J.G., Marrs, R.H., Martínez-Ruiz, C., 2012. Soil and vegetation development during early succession on restored coal wastes: a six-year permanent plot study. *Plant Soil* 353, 305–320. <https://doi.org/10.1007/s11104-011-1033-2>.
- Alignan, J.F., Debras, J.F., Dutoit, T., 2018a. Orthoptera prove good indicators of grassland rehabilitation success in the first French Natural Area Reserve. *J. Nat. Conserv.* 44, 1–11. <https://doi.org/10.1016/j.jnc.2018.04.002>.
- Alignan, J.F., Debras, J.F., Jaunatre, R., Dutoit, T., 2018b. Effects of ecological restoration on beetle assemblages: results from a large-scale experiment in a Mediterranean steppe rangeland. *Biodivers. Conserv.* 27, 2155–2172.
- Anderson, J.D., Ingram, L.J., Stahl, P.D., 2008. Influence of reclamation management practices on microbial biomass carbon and soil organic carbon accumulation in semiarid mined lands of Wyoming. *Appl. Soil Ecol.* 40, 387–397. <https://doi.org/10.1016/j.apsoil.2008.06.008>.
- Angulo-Jaramillo, R., Vandervaere, J.-P., Roulier, S., Thony, J.-L., Gaudet, J.-P., Vauclin, M., 2000. Field measurement of soil surface hydraulic properties by disc and ring infiltrometers: A review and recent developments. *Soil Tillage Res.* 55, 1–29. [https://doi.org/10.1016/S0167-1987\(00\)00098-2](https://doi.org/10.1016/S0167-1987(00)00098-2).
- Anjos, L., Gaistardo, C.C., Deckers, J., Dondeyne, S., Eberhardt, E., Gerasimova, M., Harms, B., Jones, A., Krasilnikov, P., Reinsch, T., Vargas, R., Zhang, G.-L., 2016. World reference base for soil resources 2014 International soil classification system for naming soils and creating legends for soil maps [WWW Document]. JRC Publications Repository. URL <https://publications.jrc.ec.europa.eu/repository/handle/JRC91947> (accessed 1.25.23).
- Archer, N.A.L., Quinton, J.N., Hess, T.M., 2002. Below-ground relationships of soil texture, roots and hydraulic conductivity in two-phase mosaic vegetation in South-east Spain. *J. Arid Environ.* 52, 535–553. <https://doi.org/10.1006/jare.2002.1011>.
- Archer, N.A.L., Otten, W., Schmidt, S., Bengough, A.G., Shah, N., Bonell, M., 2016. Rainfall infiltration and soil hydrological characteristics below ancient forest, planted forest and grassland in a temperate northern climate. *Ecology* 9, 585–600. <https://doi.org/10.1002/eco.1658>.
- Audet, P., Arnold, S., Lechner, A.M., Baumgartl, T., 2013. Site-specific climate analysis elucidates revegetation challenges for post-mining landscapes in eastern Australia. *Biogeosciences* 10, 6545–6557. <https://doi.org/10.5194/bg-10-6545-2013>.
- Azcárate, F.M., Peco, B., 2007. Harvester ants (*Messor barbarus*) as disturbance agents in Mediterranean grasslands. *J. Veg. Sci.* 18, 103–110. <https://doi.org/10.1111/j.1654-1103.2007.tb02520.x>.
- Baize, D., Association française pour l'étude du sol (AFES), Girard, M.-C., 2009. *Référentiel pédologique 2008*, Editions Quae, Versailles, France, pp. 1–405.
- Baize, D., Jabiou, B., 2011. *Guide pour la description des sols*. Editions Quae, Versailles, France, pp. 1–430.
- Baize, D., 2018. *Guide des analyses en pédologie 3e édition revue et augmentée*, Edition Quae, Versailles, pp. 1–328.
- Balmford, A., Bond, W., 2005. Trends in the state of nature and their implications for human well-being. *Ecol. Lett.* 8, 1218–1234. <https://doi.org/10.1111/j.1461-0248.2005.00814.x>.
- Benjamini, Y., Hochberg, Y., 1995. Controlling the False Discovery Rate: A Practical and Powerful Approach to Multiple Testing. *J. Roy. Stat. Soc. Ser. B (Methodol.)* 57, 289–300. <https://doi.org/10.1111/j.2517-6161.1995.tb02031.x>.
- Berli, M., Carminati, A., Ghezzehei, T.A., Or, D., 2008. Evolution of unsaturated hydraulic conductivity of aggregated soils due to compressive forces. *Water Resour. Res.* 44 (5) <https://doi.org/10.1029/2007WR006501>.
- Beven, K., Germann, P., 2013. Macropores and water flow in soils revisited. *Water Resour. Res.* 49, 3071–3092. <https://doi.org/10.1002/wrcr.20156>.
- Bisdorf, E.B.A., Dekker, L.W., Schoube, J.F.T., 1993. Water repellency of sieve fractions from sandy soils and relationships with organic material and soil structure. *Geoderma* 56, 105–118. [https://doi.org/10.1016/0016-7061\(93\)90103-R](https://doi.org/10.1016/0016-7061(93)90103-R).
- Bochet, E., Rubio, J.L., Poesen, J., 1999. Modified topsoil islands within patchy Mediterranean vegetation in SE Spain. *Catena* 38, 23–44. [https://doi.org/10.1016/S0341-8162\(99\)00056-9](https://doi.org/10.1016/S0341-8162(99)00056-9).

- Bonet, A., 2004. Secondary succession of semi-arid Mediterranean old-fields in south-eastern Spain: insights for conservation and restoration of degraded lands. *J. Arid Environ.* 56, 213–233. [https://doi.org/10.1016/S0140-1963\(03\)00048-X](https://doi.org/10.1016/S0140-1963(03)00048-X).
- Bottinelli, N., Jouquet, P., Capowiez, Y., Podwojewski, P., Grimaldi, M., Peng, X., 2015. Why is the influence of soil macrofauna on soil structure only considered by soil ecologists? Soil and Tillage Research, Soil Structure and its Functions in Ecosystems: Phase matter & Scale matter 146, 118–124. <https://doi.org/10.1016/j.still.2014.01.007>.
- Bouteyre, G., Duclos, G., 1994. Carte pédologique de France à 1/100 000 Arles. Carte et référentiels pédologiques.
- Buisson, E., De Almeida, T., Durbecq, A., Arruda, A.J., Vidaller, C., Alignan, J.-F., Toma, T.S.P., Hess, M.C.M., Pavon, D., Isselin-Nondedeu, F., Jaunatre, R., Moinardeau, C., Young, T.P., Mesléard, F., Dutoit, T., Blight, O., Bischoff, A., 2021. Key issues in Northwestern Mediterranean dry grassland restoration. *Restor. Ecol.* 29, e13258 <https://doi.org/10.1111/rec.13258>.
- Buisson, E., Dutoit, T., 2006. Creation of the natural reserve of La Crau: Implications for the creation and management of protected areas. *J. Environ. Manage.* 80, 318–326. <https://doi.org/10.1016/j.jenvman.2005.09.013>.
- Buisson, E., Dutoit, T., Rolando, C., 2004. Composition et structure de la végétation aux bordures entre friches post-culturales et végétation steppique dans la plaine de Crau (Bouches-du-Rhône). *Ecologia mediterranea* 30, 71–84.
- Buisson, E., Jaunatre, R., Römermann, C., Bulot, A., Dutoit, T., 2018. Species transfer via topsoil translocation: lessons from two large Mediterranean restoration projects. *Restor. Ecol.* 26, S179–S188. <https://doi.org/10.1111/rec.12682>.
- Bullock, J.M., 1998. Community translocation in Britain: Setting objectives and measuring consequences. *Biol. Conserv.* 84, 199–214. [https://doi.org/10.1016/S0006-3207\(97\)00140-7](https://doi.org/10.1016/S0006-3207(97)00140-7).
- Bulot, A., Dutoit, T., Renucci, M., Provost, E., 2014a. A new protocol for harvester ant queens *Messor barbarus* (Hymenoptera: Formicidae) to improve the restoration of species-rich plant communities in the future. *Myrmecol. News.* 20, 43–52.
- Bulot, A., Provost, E., Dutoit, T., 2014b. A comparison of different soil transfer strategies for restoring a Mediterranean steppe after a pipeline leak (La Crau plain, South-Eastern France). *Ecol. Eng.* 71, 690–702. <https://doi.org/10.1016/j.ecoleng.2014.07.060>.
- Bulot, A., Provost, E., Dutoit, T., 2016. Refuse pile turnover by harvester ants (Hymenoptera: Formicidae) increases seed density and seedling species richness in dry grasslands. *Myrmecological News* 23, 91–100.
- Bulot, A., Potard, K., Bureau, F., Bérard, A., Dutoit, T., 2017. Ecological restoration by soil transfer: impacts on restored soil profiles and topsoil functions. *Restor. Ecol.* 25, 354–366. <https://doi.org/10.1111/rec.12424>.
- Capowiez, Y., Sammartino, S., Keller, T., Bottinelli, N., 2021. Decreased burrowing activity of endogeic earthworms and effects on water infiltration in response to an increase in soil bulk density. *Pedobiologia* 85–86, 150728. <https://doi.org/10.1016/j.pedobi.2021.150728>.
- Cerdà, A., 1997. Seasonal changes of the infiltration rates in a Mediterranean scrubland on limestone. *J. Hydrol.* 198, 209–225. [https://doi.org/10.1016/S0022-1694\(96\)03295-7](https://doi.org/10.1016/S0022-1694(96)03295-7).
- Chenot, J., Jaunatre, R., Buisson, E., Dutoit, T., 2017. Long-term effects of topsoil transfer assessed thirty years after rehabilitation of dry alluvial quarries in Southeastern France. *Ecol. Eng.* 99, 1–12. <https://doi.org/10.1016/j.ecoleng.2016.11.010>.
- Chenot, J., Jaunatre, R., Buisson, E., Bureau, F., Dutoit, T., 2018. Impact of quarry exploitation and disuse on pedogenesis. *Catena* 160, 354–365. <https://doi.org/10.1016/j.catena.2017.09.012>.
- Chessel, D., Dufour, A.B., Thioulouse, J., 2004. The ade4 package—One-table methods. *R news* 4, 5–10.
- Clewell, A.F., Aronson, J., 2013. *Ecological Restoration, Second Edition: Principles, Values and Structure of an Emerging Profession*, ed. Island Press.
- Coiffait-Gombault, C., Buisson, E., Dutoit, T., 2011. Hay Transfer Promotes Establishment of Mediterranean Steppe Vegetation on Soil Disturbed by Pipeline Construction. *Restoration Ecol.* 19, 214–222. <https://doi.org/10.1111/j.1526-100X.2010.00706.x>.
- Coiffait-Gombault, C., Buisson, E., Dutoit, T., 2012. Are old Mediterranean grasslands resilient to human disturbances? *Acta Oecol.* 43, 86–94. <https://doi.org/10.1016/j.actao.2012.04.011>.
- Colloff, M.J., Pullen, K.R., Cunningham, S.A., 2010. Restoration of an Ecosystem Function to Revegetation Communities: The Role of Invertebrate Macropores in Enhancing Soil Water Infiltration. *Restor. Ecol.* 18, 65–72. <https://doi.org/10.1111/j.1526-100X.2010.00667.x>.
- Crockford, H., Topalidis, S., Richardson, D.P., 1991. Water repellency in a dry sclerophyll eucalypt forest — measurements and processes. *Hydrol. Process.* 5, 405–420. <https://doi.org/10.1002/hyp.3360050408>.
- Dane, J.H., Hopmans, J.W., 2002. 3.3.2 Laboratory, n: Dane, J.H., Topp, G.C. (Eds.), *Methods of Soil Analysis: Part 4 Physical Methods*. Soil Science Society of America, Inc., Madison, WI. 10.2136/sssabookser5.4.c25.
- De Almeida, T., Blight, O., Mesléard, F., Bulot, A., Provost, E., Dutoit, T., 2020. Harvester ants as ecological engineers for Mediterranean grassland restoration: Impacts on soil and vegetation. *Biol. Conserv.* 245, 108547 <https://doi.org/10.1016/j.biocon.2020.108547>.
- De Deyn, G.B., Raaijmakers, C.E., Zoomer, H.R., Berg, M.P., de Ruiter, P.C., Verhoef, H.A., Bezemer, T.M., van der Putten, W.H., 2003. Soil invertebrate fauna enhances grassland succession and diversity. *Nature* 422, 711–713. <https://doi.org/10.1038/nature01548>.
- Deng, L., Wang, K., Li, J., Zhao, G., Shangquan, Z., 2016. Effect of soil moisture and atmospheric humidity on both plant productivity and diversity of native grasslands across the Loess Plateau, China. *Ecol. Eng.* 94, 525–531. <https://doi.org/10.1016/j.ecoleng.2016.06.048>.
- DePuit, E.J., 1984. Potential topsoiling strategies for enhancement of vegetation diversity on mined lands. *Minerals and the Environment* 6, 115–120. <https://doi.org/10.1007/BF02043991>.
- Deurer, M., Müller, K., Van Den Dijssel, C., Mason, K., Carter, J., Clothier, B.E., 2011. Is soil water repellency a function of soil order and proneness to drought? A survey of soils under pasture in the North Island of New Zealand. *Eur. J. Soil Sci.* 62, 765–779. <https://doi.org/10.1111/j.1365-2389.2011.01392.x>.
- Devaux, J.P., Archiloque, A., Borel, L., Bourrelly, M., Louis-Palluel, J., 1983. Notice de la carte phyto-écologique de la Crau (Bouches du Rhône). *Biologie et écologie méditerranéenne* 10, 5–54.
- Doerr, S.H., 1998. On standardizing the Water Drop Penetration Time and the Molarity of an Ethanol Droplet techniques to classify soil hydrophobicity: A case study using medium textured soils. *Earth Surf. Proc. Land.* 23 (7), 663–668. [https://doi.org/10.1002/\(SICI\)1096-9837\(199807\)23:7<663::AID-ESP909>3.0.CO;2-6](https://doi.org/10.1002/(SICI)1096-9837(199807)23:7<663::AID-ESP909>3.0.CO;2-6).
- Dray, S., Dufour, A.-B., 2007. The ade4 package: implementing the duality diagram for ecologists. *J. Stat. Softw.* 22, 1–20. <https://doi.org/10.18637/jss.v022.i04>.
- Dutoit, T., Buisson, E., Fadda, S., Henry, F., Coiffait-Gombault, C., Jaunatre, R., Alignan, J.-F., Masson, S., Bulot, A., 2013. The pseudo-steppe of La Crau (South-Eastern France): origin, management and restoration of a Mediterranean rangeland. In: Traba, J., Morales, M. (Eds.), *Steppe Ecosystems: Biological Diversity, Management and Restoration*, 2nd ed. Nova Publishers, Hauppauge (NY), pp. 287–301.
- Dzwonko, Z., Loster, S., 1998. Dynamics of species richness and composition in a limestone grassland restored after tree cutting. *J. Veg. Sci.* 9, 387–394. <https://doi.org/10.2307/3237103>.
- Edwards, C.A., 2002. Assessing the effects of environmental pollutants on soil organisms, communities, processes and ecosystems. *Eur. J. Soil Biol.* 38, 225–231. [https://doi.org/10.1016/S1164-5563\(02\)01150-0](https://doi.org/10.1016/S1164-5563(02)01150-0).
- Eldridge, D.J., Bowker, M.A., Maestre, F.T., Alonso, P., Mau, R.L., Papadopoulos, J., Escudero, A., 2010. Interactive Effects of Three Ecosystem Engineers on Infiltration in a Semi-Arid Mediterranean Grassland. *Ecosystems* 13, 499–510. <https://doi.org/10.1007/s10021-010-9335-4>.
- Enright, N.J., Miller, B.P., Akhter, R., 2005. Desert vegetation and vegetation-environment relationships in Kirthar National Park, Sindh, Pakistan. *J. Arid Environ.* 61, 397–418. <https://doi.org/10.1016/j.jaridenv.2004.09.009>.
- Ernst, G., Felten, D., Vohland, M., Emmerling, C., 2009. Impact of ecologically different earthworm species on soil water characteristics. *Eur. J. Soil Biol.* 45, 207–213. <https://doi.org/10.1016/j.ejsobi.2009.01.001>.
- Etna, A., Håkansson, I., 1994. Swedish experiments on the persistence of subsoil compaction caused by vehicles with high axle load. *Soil and Tillage Research, Subsoil Compaction by High Axle Load Traffic* 29, 167–172. [https://doi.org/10.1016/0167-1987\(94\)90053-1](https://doi.org/10.1016/0167-1987(94)90053-1).
- Falipou, P., Legros, J.P., 2002. The STIPA-2000 system conceived to enter and display the soil data stored in the national base of France (DONESOL.II). *Etude et Gestion des Sols* 9, 55.
- Fischer, C., Roscher, C., Jensen, B., Eisenhauer, N., Baade, J., Attinger, S., Scheu, S., Weisser, W.W., Schumacher, J., Hildebrandt, A., 2014. How Do Earthworms, Soil Texture and Plant Composition Affect Infiltration along an Experimental Plant Diversity Gradient in Grassland? *PLoS One* 9, e98987. <https://doi.org/10.1371/journal.pone.0098987>.
- Fischer, C., Tischer, J., Roscher, C., Eisenhauer, N., Ravenek, J., Gleixner, G., Attinger, S., Jensen, B., de Kroon, H., Mommer, L., Scheu, S., Hildebrandt, A., 2015. Plant species diversity affects infiltration capacity in an experimental grassland through changes in soil properties. *Plant Soil* 397, 1–16. <https://doi.org/10.1007/s11104-014-2373-5>.
- Frouz, J., Prach, K., Pižl, V., Háníl, L., Starý, J., Tajovský, K., Materna, J., Balík, V., Kalcík, J., Řehounková, K., 2008. Interactions between soil development, vegetation and soil fauna during spontaneous succession in post mining sites. *Eur. J. Soil Biol.* 44, 109–121. <https://doi.org/10.1016/j.ejsobi.2007.09.002>.
- Gann, G.D., McDonald, T., Aronson, J., Dixon, K.W., Walder, B., Hallett, J.G., Decler, K., Falk, D.A., Gonzales, E.K., Murcia, C., Nelson, C.R., Unwin, A.J., 2018. The SER Standards: a globally relevant and inclusive tool for improving restoration practice—a reply to Higgs. et al. *Restor. Ecol.* 26, 426–430. <https://doi.org/10.1111/rec.12819>.
- Gibson, D.J., Gibson, D.J., 2008. *Grasses and Grassland Ecology*. Oxford University Press, Oxford, New York.
- Golos, P.J., Dixon, K.W., Erickson, T.E., 2016. Plant recruitment from the soil seed bank depends on topsoil stockpile age, height, and storage history in an arid environment. *Restor. Ecol.* 24, S53–S61. <https://doi.org/10.1111/rec.12389>.
- Gould, I.J., Quinton, J.N., Weigelt, A., De Deyn, G.B., Bardgett, R.D., 2016. Plant diversity and root traits benefit physical properties key to soil function in grasslands. *Ecol. Lett.* 19, 1140–1149. <https://doi.org/10.1111/ele.12652>.
- Habel, J.C., Dengler, J., Janišová, M., Török, P., Wellstein, C., Wiezik, M., 2013. European grassland ecosystems: threatened hotspots of biodiversity. *Biodivers Conserv* 22, 2131–2138. <https://doi.org/10.1007/s10531-013-0537-x>.
- Hallam, J., Hodson, M.E., 2020. Impact of different earthworm ecotypes on water stable aggregates and soil water holding capacity. *Biol Fertil Soils* 56, 607–617. <https://doi.org/10.1007/s00374-020-01432-5>.
- Hamza, M.A., Anderson, W.K., 2005. Soil compaction in cropping systems: A review of the nature, causes and possible solutions. *Soil Tillage Res.* 82, 121–145. <https://doi.org/10.1016/j.still.2004.08.009>.
- Harris, J.A., Hobbs, R.J., Higgs, E., Aronson, J., 2006. Ecological Restoration and Global Climate Change. *Restor. Ecol.* 14, 170–176. <https://doi.org/10.1111/j.1526-100X.2006.00136.x>.

- Haverkamp, R., Arrúe, J.L., Vandervaere, J.P., Braud, I., Boulet, G., Laurent, J.P., Taha, A., Ross, P.J., Angulo-Jaramillo, R., 1996. Hydrological and thermal behaviour of the vadose zone in the area of Barrax and Tomelloso (Spain): Experimental study, analysis and modeling. *Project UE n. EV5C-CT 92*, 00–90.
- Henry, F., Talon, B., Dutoit, T., 2010. The age and history of the French Mediterranean steppe revisited by soil wood charcoal analysis. *The Holocene* 20, 25–34. <https://doi.org/10.1177/0959683609348841>.
- Janišová, M., Bartha, S., Kiehl, K., Dengler, J., 2011. Advances in the conservation of dry grasslands: Introduction to contributions from the seventh European Dry Grassland Meeting. *Plant Biosyst.* 145, 507–513. <https://doi.org/10.1080/11263504.2011.603895>.
- Jarvis, N.J., Zavattaro, L., Rajkai, K., Reynolds, W.D., Olsen, P.-A., McGechan, M., Mecke, M., Mohanty, B., Leeds-Harrison, P.B., Jacques, D., 2002. Indirect estimation of near-saturated hydraulic conductivity from readily available soil information. *Geoderma* 108, 1–17. [https://doi.org/10.1016/S0016-7061\(01\)00154-9](https://doi.org/10.1016/S0016-7061(01)00154-9).
- Jasper, D.A., 2007. Beneficial Soil Microorganisms of the Jarrah Forest and Their Recovery in Bauxite Mine Restoration in Southwestern Australia. *Restor. Ecol.* 15, S74–S84. <https://doi.org/10.1111/j.1526-100X.2007.00295.x>.
- Jaumatre, R., Buisson, E., Dutoit, T., 2014. Can ecological engineering restore Mediterranean rangeland after intensive cultivation? A large-scale experiment in southern France. *Ecol. Eng.* 64, 202–212. <https://doi.org/10.1016/j.ecoeng.2013.12.022>.
- Jouquet, P., Dauber, J., Lagerlöf, J., Lavelle, P., Lepage, M., 2006. Soil invertebrates as ecosystem engineers: Intended and accidental effects on soil and feedback loops. *Appl. Soil Ecol.* 32, 153–164. <https://doi.org/10.1016/j.apsoil.2005.07.004>.
- Khlosi, M., Cornelis, W.M., Douaik, A., Hazzouri, A., Habib, H., Gabriels, D., 2013. Exploration of the Interaction between Hydraulic and Physicochemical Properties of Syrian Soils. *Vadose Zone J.* 12 (vzj2012), 0209. <https://doi.org/10.2136/vzj2012.0209>.
- Kiehl, K., Kirmer, A., Donath, T.W., Rasan, L., Hölzel, N., 2010. Species introduction in restoration projects – Evaluation of different techniques for the establishment of semi-natural grasslands in Central and Northwestern Europe. *Basic Appl. Ecol.* 11, 285–299. <https://doi.org/10.1016/j.baec.2009.12.004>.
- King, P., 1981. Comparison of Methods for Measuring Severity of Water Repellence of Sandy Soils and Assessment of Some Factors That Affect Its Measurement. *Aust. J. Soil Res.* 19, 275–285. <https://doi.org/10.1071/SR9810275>.
- Kottek, M., Grieser, J., Beck, C., Rudolf, B., Rubel, F., 2006. World map of the Köppen-Geiger climate classification updated. *Meteorol. Z.* 15, 259–263. <https://doi.org/10.1127/0941-2948/2006/0130>.
- Lamb, D., Erskine, P.D., Fletcher, A., 2015. Widening gap between expectations and practice in Australian minesite rehabilitation. *Ecol. Manag. Restor.* 16, 186–195. <https://doi.org/10.1111/emr.12179>.
- Lassabatère, L., Angulo-Jaramillo, R., Soria Ugalde, J.M., Cuenca, R., Braud, I., Haverkamp, R., 2006. Beerkan Estimation of Soil Transfer Parameters through Infiltration Experiments—BEST. *Soil Sci. Soc. Am. J.* 70, 521–532. <https://doi.org/10.2136/sssaj2005.0026>.
- Lavelle, P., 1997. Faunal Activities and Soil Processes: Adaptive Strategies That Determine Ecosystem Function. In: Begon, M., Fitter, A.H. (Eds.), *Advances in Ecological Research*. Academic Press, pp. 93–132. [https://doi.org/10.1016/S0065-2504\(08\)60007-0](https://doi.org/10.1016/S0065-2504(08)60007-0).
- Lee, K.E., Foster, R.C., 1991. Soil fauna and soil structure. *Soil Res.* 29, 745–775. <https://doi.org/10.1071/sr9910745>.
- Lee, T.D., Tjoelker, M.G., Ellsworth, D.S., Reich, P.B., 2001. Leaf gas exchange responses of 13 prairie grassland species to elevated CO₂ and increased nitrogen supply. *New Phytol.* 150, 405–418. <https://doi.org/10.1046/j.1469-8137.2001.00095.x>.
- Lipiec, J., Wójciga, A., Horn, R., 2009. Hydraulic properties of soil aggregates as influenced by compaction. *Soil Tillage Res.* 103, 170–177. <https://doi.org/10.1016/j.still.2008.10.021>.
- Liu, Y., Cui, Z., Huang, Z., López-Vicente, M., Wu, G.-L., 2019a. Influence of soil moisture and plant roots on the soil infiltration capacity at different stages in arid grasslands of China. *Catena* 182, 104147. <https://doi.org/10.1016/j.catena.2019.104147>.
- Liu, Y., Miao, H.-T., Chang, X., Wu, G.-L., 2019b. Higher species diversity improves soil water infiltration capacity by increasing soil organic matter content in semiarid grasslands. *Land Degrad. Dev.* 30, 1599–1606. <https://doi.org/10.1002/ldr.3349>.
- Lobry de Bruyn, L.A., Conacher, A.J., 1994. The effect of ant biopores on water infiltration in soils in undisturbed bushland and in farmland in a semi-arid environment. *Pedobiologia.* 38, 193–207.
- Loch, R.J., 2000. Effects of vegetation cover on runoff and erosion under simulated rain and overland flow on a rehabilitated site on the Meandu Mine, Tarong, Queensland. *Soil Res.* 38, 299–312. <https://doi.org/10.1071/sr99030>.
- Ludwig, J.A., Wilcox, B.P., Breshers, D.D., Tongway, D.J., Imeson, A.C., 2005. Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. *Ecology* 86 (2), 288–297. <https://doi.org/10.1890/03-0569>.
- Maestre, F.T., Cortina, J., 2002. Spatial patterns of surface soil properties and vegetation in a Mediterranean semi-arid steppe. *Plant Soil* 241, 279–291. <https://doi.org/10.1023/A:1016172308462>.
- Martin, G., Courtial, A., Génin, A., Ramone, H., Dutoit, T., 2022. Why grazing and soil matter for dry grassland diversity: New insights from multigroup structural equation modeling of micro-patterns. *Front. Ecol. Evol.* 10, 879060. <https://doi.org/10.3389/fevo.2022.879060>.
- Martín-Perera, D., Fesharaki, O., Domingo, M.S., Gamboa, S., Hernández Fernández, M., 2019. Messor barbarus ants as soil bioturbators: Implications for granulometry, mineralogical composition and fossil remains extraction in Somoasguas site (Madrid basin, Spain). *Catena* 172, 664–677. <https://doi.org/10.1016/j.catena.2018.09.018>.
- McBride, R.A., Mackintosh, E.E., 1984. Soil Survey Interpretations from Water Retention Data: I. Development and Validation of a Water Retention Model. *Soil Sci. Soc. Am. J.* 48, 1338–1343. <https://doi.org/10.2136/sssaj1984.03615995004800060028x>.
- McKenzie, D., Gedalof, Z., Peterson, D.L., Mote, P., 2004. Climatic Change, Wildfire, and Conservation. *Conserv. Biol.* 18, 890–902. <https://doi.org/10.1111/j.1523-1739.2004.00492.x>.
- Meek, B.D., Rechel, E.R., Carter, L.M., DeTar, W.R., Urie, A.L., 1992. Infiltration Rate of a Sandy Loam Soil: Effects of Traffic, Tillage, and Plant Roots. *Soil Sci. Soc. Am. J.* 56, 908–913. <https://doi.org/10.2136/sssaj1992.03615995005600030038x>.
- Metson, A.J., 1957. Methods of Chemical Analysis for Soil Survey Samples. *Soil Sci.* 83, 245.
- Michaelides, K., Lister, D., Wainwright, J., Parsons, A.J., 2009. Vegetation controls on small-scale runoff and erosion dynamics in a degrading dryland environment. *Hydrol. Process* 23 (11), 1617–1630. <https://doi.org/10.1002/hyp.7293>.
- Mohanty, B.P., Bowman, R.S., Hendrickx, J.M.H., van Genuchten, M.T., 1997. New piecewise-continuous hydraulic functions for modeling preferential flow in an intermittent-flood-irrigated field. *Water Resour. Res.* 33, 2049–2063. <https://doi.org/10.1029/97WR01701>.
- Molinier, R., Tallon, G., 1950. La végétation de la Crau (Basse-Provence). *Rev. Gen. Bot.* 56, 1–111.
- Molliex, S., Siame, L.L., Bourlès, D.L., Bellier, O., Braucher, R., Clauzon, G., 2013. Quaternary evolution of a large alluvial fan in a periglacial setting (Crau Plain, SE France) constrained by terrestrial cosmogenic nuclide (10Be). *Geomorphology* 195, 45–52. <https://doi.org/10.1016/j.geomorph.2013.04.025>.
- Olsen, S.R., 1954. Estimation of Available Phosphorus in Soils by Extraction with Sodium Bicarbonate. U.S. Department of Agriculture.
- Piqueray, J., Bottin, G., Delescaille, L.-M., Bisteau, E., Colinet, G., Mahy, G., 2011. Rapid restoration of a species-rich ecosystem assessed from soil and vegetation indicators: The case of calcareous grasslands restored from forest stands. *Ecol. Ind.* 11, 724–733. <https://doi.org/10.1016/j.ecoind.2010.06.007>.
- Piqueray, J., Gilliaux, V., Wubs, E.R.J., Mahy, G., 2020. Topsoil translocation in extensively managed arable field margins promotes plant species richness and threatened arable plant species. *J. Environ. Manage.* 260, 110126. <https://doi.org/10.1016/j.jenvman.2020.110126>.
- Poschold, P., WallisDeVries, M.F., 2002. The historical and socioeconomic perspective of calcareous grasslands—lessons from the distant and recent past. *Biol. Conserv.* 104, 361–376. [https://doi.org/10.1016/S0006-3207\(01\)00201-4](https://doi.org/10.1016/S0006-3207(01)00201-4).
- Pueyo, Y., Moret-Fernández, D., Saiz, H., Bueno, C.G., Alados, C.L., 2013. Relationships Between Plant Spatial Patterns, Water Infiltration Capacity, and Plant Community Composition in Semi-arid Mediterranean Ecosystems Along Stress Gradients. *Ecosystems* 16, 452–466. <https://doi.org/10.1007/s10021-012-9620-5>.
- Puigdefábregas, J., 2005. The role of vegetation patterns in structuring runoff and sediment fluxes in drylands. *Earth Surf. Process. Land.* 30, 133–147. <https://doi.org/10.1002/esp.1181>.
- Queiroz, C., Beilin, R., Folke, C., Lindborg, R., 2014. Farmland abandonment: threat or opportunity for biodiversity conservation? A global review. *Front. Ecol. Environ.* 12, 288–296. <https://doi.org/10.1890/120348>.
- Radford, B.J., Wilson-Rummenie, A.C., Simpson, G.B., Bell, K.L., Ferguson, M.A., 2001. Compacted soil affects soil macrofauna populations in a semi-arid environment in central Queensland. *Soil Biol. Biochem.* 33, 1869–1872. [https://doi.org/10.1016/S0038-0717\(01\)00104-3](https://doi.org/10.1016/S0038-0717(01)00104-3).
- Reynolds, H.L., Packer, A., Bever, J.D., Clay, K., 2003. Grassroots Ecology: Plant–Microbe–Soil Interactions as Drivers of Plant Community Structure and Dynamics. *Ecology* 84, 2281–2291. <https://doi.org/10.1890/02-0298>.
- Saatkamp, A., Römermann, C., Dutoit, T., 2010. Plant Functional Traits Show Non-Linear Response to Grazing. *Folia Geobot.* 45, 239–252. <https://doi.org/10.1007/s12224-010-9069-2>.
- Sanchez, C.E., Karl Wood, M., 1989. Infiltration rates and erosion associated with reclaimed coal mine spoils in west central New Mexico. *Landscape and Urban Planning, Special Issue Reclamation and Revegetation* 17, 151–168. [https://doi.org/10.1016/0169-2046\(89\)90023-6](https://doi.org/10.1016/0169-2046(89)90023-6).
- Sheoran, V., Sheoran, A.S., Poonia, P., 2010. Soil reclamation of abandoned mine land by revegetation: a review. *International journal of soil, sediment and water* 3 (2), 13.
- Thompson, S.E., Harman, C.J., Heine, P., Katul, G.G., 2010. Vegetation-infiltration relationships across climatic and soil type gradients. *J. Geophys. Res. Biogeophys.* 115. <https://doi.org/10.1029/2009JG001134>.
- Török, P., Vida, E., Deák, B., Lengyel, S., Tóthmérész, B., 2011. Grassland restoration on former croplands in Europe: an assessment of applicability of techniques and costs. *Biodivers Conserv* 20, 2311–2332. <https://doi.org/10.1007/s10531-011-9992-4>.
- Traba, J., Levassor, C., Peco, B., 2003. Restoration of Species Richness in Abandoned Mediterranean Grasslands: Seeds in Cattle Dung. *Restor. Ecol.* 11, 378–384. <https://doi.org/10.1046/j.1526-100X.2003.00227.x>.
- Unger, P.W., Kaspar, T.C., 1994. Soil Compaction and Root Growth: A Review. *Agron. J.* 86, 759–766. <https://doi.org/10.2134/agronj1994.00021962008600050004x>.
- van Genuchten, M.T., 1980. A Closed-form Equation for Predicting the Hydraulic Conductivity of Unsaturated Soils. *Soil Sci. Soc. Am. J.* 44, 892–898. <https://doi.org/10.2136/sssaj1980.03615995004400050002x>.
- Vereecken, H., 1995. Estimating the unsaturated hydraulic conductivity from theoretical models using simple soil properties. *Geoderma* 65, 81–92. [https://doi.org/10.1016/0016-7061\(95\)92543-X](https://doi.org/10.1016/0016-7061(95)92543-X).
- Vereecken, H., Weynants, M., Javaux, M., Pachepsky, Y., Schaap, M.G., van Genuchten, M.T., 2010. Using Pedotransfer Functions to Estimate the van Genuchten-Mualem Soil Hydraulic Properties: A Review. *Vadose Zone J.* 9, 795–820. <https://doi.org/10.2136/vzj2010.0045>.

- Watson, C.L., Letey, J., 1970. Indices for Characterizing Soil-Water Repellency Based upon Contact Angle-Surface Tension Relationships. *Soil Sci. Soc. Am. J.* 34, 841–844. <https://doi.org/10.2136/sssaj1970.03615995003400060011x>.
- Weiher, E., Keddy, P. (Eds.), 1999. *Ecological Assembly Rules: Perspectives, Advances, Retreats*. Cambridge University Press, Cambridge. 10.1017/CBO9780511542237.
- Weng, E., Luo, Y., 2008. Soil hydrological properties regulate grassland ecosystem responses to multifactor global change: A modeling analysis. *J. Geophys. Res. Biogeo.* 113 <https://doi.org/10.1029/2007JG000539>.
- White, R., 2000. Pilot analysis of global ecosystems: Grassland ecosystems.
- Willems, J.H., 2001. Problems, Approaches, and Results in Restoration of Dutch Calcareous Grassland During the Last 30 Years. *Restor. Ecol.* 9, 147–154. <https://doi.org/10.1046/j.1526-100x.2001.009002147.x>.
- Williams, N.S.G., Schwartz, M.W., Vesk, P.A., McCarthy, M.A., Hahs, A.K., Clemants, S. E., Corlett, R.T., Duncan, R.P., Norton, B.A., Thompson, K., McDonnell, M.J., 2009. A conceptual framework for predicting the effects of urban environments on floras. *J. Ecol.* 97, 4–9. <https://doi.org/10.1111/j.1365-2745.2008.01460.x>.
- Wills, B.D., Landis, D.A., 2018. The role of ants in north temperate grasslands: a review. *Oecologia* 186, 323–338. <https://doi.org/10.1007/s00442-017-4007-0>.
- Wösten, J.H.M., Pachepsky, Ya.A., Rawls, W.J., 2001. Pedotransfer functions: bridging the gap between available basic soil data and missing soil hydraulic characteristics. *J. Hydrol.* 251, 123–150. [https://doi.org/10.1016/S0022-1694\(01\)00464-4](https://doi.org/10.1016/S0022-1694(01)00464-4).
- WRB, 2015. *World Reference Base for Soil Resources 2014. Update 2015. World Soil Resources Reports No. 106*, n.d. Food and Agriculture Organization of the United Nations, Rome, pp. 192–p.
- Wu, G.-L., Zhang, Z.-N., Wang, D., Shi, Z.-H., Zhu, Y.-J., 2014. Interactions of soil water content heterogeneity and species diversity patterns in semi-arid steppes on the Loess Plateau of China. *J. Hydrol.* 519, 1362–1367. <https://doi.org/10.1016/j.jhydrol.2014.09.012>.
- Wubs, E.R.J., van der Putten, W.H., Bosch, M., Bezemer, T.M., 2016. Soil inoculation steers restoration of terrestrial ecosystems. *Nat. Plants* 2, 1–5. <https://doi.org/10.1038/nplants.2016.107>.