



More is not necessarily better: The role of cover and spatial organization of resource sinks in the restoration of patchy drylands

Paul M.J. Berghuis^{*}, Ángeles G. Mayor, Max Rietkerk, Mara Baudena

Copernicus Institute of Sustainable Development, Utrecht University, PO Box 80115, 3508 TC, Utrecht, the Netherlands

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ABSTRACT

A low-cost restoration action in patchy drylands worldwide is the installation of obstructions (hereafter, resource sinks) to break runoff pathways and retain resources. Field-works have studied how the effectiveness of this action depends on the materials installed. However, the influence and effectiveness of the cover and spatial organization of resource sinks have not been widely investigated. In this work, we use a well-known dryland model to study how different initial cover and spatial organization of installed resource sinks affect the recovered ecosystem. In agreement with field-work studies, our results confirm that the installation of resource sinks can restore degraded drylands that would not recover naturally. More importantly, a very small cover of resource sinks was sufficient to trigger vegetation recovery, while a high cover would lead to complete failure. Higher plant densities were reached when distribution and cover were similar to that of spatial self-organized vegetation in the reference healthy system (i.e., regular spatial distribution in our study system). Given the effectiveness of low cover installations, suggested by our work, combined with the low-cost materials needed, resource sinks have the potential to be a key contributor to the large restoration efforts needed to achieve land-degradation neutrality, particularly in developing countries.

1. Introduction

Halting land degradation and restoring degraded ecosystems in drylands is one of the main challenges for sustainable development (UN DESA, 2018). In recognition of this challenge, the UN has declared 2021–2030 the Decade on Ecosystem Restoration, aiming to massively scale up the restoration of degraded and destroyed ecosystems as a proven measure to fight climate change, enhance food security, water supply and biodiversity. Recent estimates suggest that restoration rates would need to exceed land degradation rates by one third to achieve land degradation neutrality aimed by the UNCCD (Ye et al., 2019). This need for widespread restoration calls for an improved efficiency of restoration actions. This is particularly true for simple low-cost restoration options that can be applied in developing countries, as these are the world's regions most affected by land degradation and climate change (Bathiany et al., 2018; Práválie, 2016).

One such simple low-cost restoration action in patchy drylands is the installation of resource sinks in the degraded target area. The main goal of this option is to recover the ability of the ecosystem to capture and store water and nutrients, by installing obstructions that break runoff

pathways and retain these resources. In this way, the positive feedback loops between vegetation loss and resource loss that underly most desertification processes and that keep the system resilient to restoration, can change in direction and foster ecosystem recovery (i.e., from vegetation loss and resource loss to vegetation gain and resource gain) (Mayor et al., 2013; Suding et al., 2004). Several field experiments in patchy drylands worldwide including Acacia woodlands in Australia and Kenya, and xerophytic Chaco forests in Argentina, have tested the efficiency of this restoration action (Cavallero et al., 2019; Kimiti et al., 2017; Tongway and Ludwig, 2011). All the experiments took place in degraded areas that had not shown recovery signs for decades. Resource sinks in these studies were built in different ways, mainly using piles of locally available branches and woody debris, sometimes accompanied with seeds, litter and/or nutrients, and/or reinforced with burlap sacking or nylon mesh. In all cases, the installed resource sinks retained sediments and nutrients, and increased water infiltration, facilitating favourable habitats for the establishment of perennial vegetation, as well as for several groups of macroinvertebrates such as ants and termites. Furthermore, Kimiti et al. (2017) provided figures demonstrating the low cost of this restoration action at the management scale. These

^{*} Corresponding author.

E-mail address: p.m.j.berghuis@uu.nl (P.M.J. Berghuis).

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experimental field works showed that the ability of the installed resource sinks to retain runoff and sediments varied depending on the materials used in their construction. However, the spatial organization of a given resource sink can also affect the effectiveness of the restoration action, as much as the spatial organization of vegetation controls the ability of a dryland landscape to retain resources (e.g., [Bautista et al., 2007](#); [McCallum et al., 2018](#)).

Nevertheless, the existing works have primarily focused on studying the effectiveness of different types of resource sinks, paying much less attention to aspects related to the spatial design of the restoration. Spatially-explicit mathematical models of drylands are particularly suitable for this task. However, despite the immense potential to evaluate and predict dryland dynamics shown by these models in the last two decades (e.g., [Kéfi et al., 2007](#); [Mayor et al., 2019](#); [Gilad et al., 2004](#); [Rietkerk et al., 2002](#), see [Borgogno et al., 2009](#) and [Meron, 2012](#) for overviews), modelling efforts have only rarely been directed to investigate dryland restoration ([James and Carrick, 2016](#); [Saco and Moreno-de las Heras, 2013](#)). In this study, we aim at developing this modelling potential and at contributing to knowledge on the spatial design of restoration by using a well-known dryland model to investigate the role of spatial organization of installed resource sinks in the restoration of degraded drylands. In particular, we investigated how different initial amounts of cover and spatial distribution of installed resource sinks (i.e., random or regular) affected the density and spatial pattern of vegetation at steady state.

2. Methods

2.1. The model

For the present study, we modified the model of [Rietkerk et al. \(2002\)](#); hereafter also referred to as “the original model”) for vegetation dynamics in a water-limited ecosystem, to include infiltration-enhancing structures (hereafter, resource sinks) in order to mimic a restoration action. This model was chosen as it is widely studied and represents well dryland ecosystems where the cover of vegetation forms self-organized patterns, capturing the essential mechanism of increased surface water infiltration with increased vegetation cover ([Rietkerk et al., 2002](#)).

The model represents the spatial dynamics of vegetation over a flat two-dimensional domain of 200×200 grid cells, where a cell has dimension $2 \text{ m} \times 2 \text{ m}$. The model describes the continuous time dynamics of three state variables: surface water (O ; mm), soil water (W ; mm) and plant density (P ; g m^{-2}). The dynamics of these state variables are described by the following three partial differential equations:

$$\frac{\partial P}{\partial t} = c g_{\max} \frac{W}{W + k_1} P - dP + P_{\text{seed}} + D_p \Delta P, \quad (1a)$$

$$\frac{\partial W}{\partial t} = \alpha O \frac{P + k_2 W_{\text{inf}}}{P + k_2} - g_{\max} \frac{W}{W + k_1} P - r_w W + D_w \Delta W, \quad (1b)$$

$$\frac{\partial O}{\partial t} = R - \alpha O \frac{P + k_2 W_{\text{inf}}}{P + k_2} + D_o \Delta O. \quad (1c)$$

W , O and P interact and affect one another by water flows and by changing several properties. Plant growth and soil water uptake are dependent on plant density (P ; linearly) and on soil water (W ; with a saturation function), see first term on the right-hand side (r.h.s.) in 1a and second in 1b. Infiltration rates increase with plant density, asymptotically reaching a maximum ([Rietkerk and van de Koppel, 1997](#)), this in turn increases soil water (first term on the r.h.s. in eq. (1b)) and decreases overland flow (second term on the r.h.s in eq. (1c)). Vegetation decay (second term on the r.h.s., eq. (1a)) and water losses (third term on the r.h.s. in eq. (1b)) are linear terms. $D_p \Delta P$, $D_w \Delta W$ and $D_o \Delta O$ (last terms in eq. 1a-c) represent diffusion terms, where Δ is the Laplacian operator for the horizontal surface coordinates. For the name and values of all parameters used see [Table 1](#), and for further details [Rietkerk et al.](#)

(2002). Important parameters for this study are rainfall (R ; mm d^{-1} ; first term on the r.h.s. in eq. (1c)), determining water availability, and infiltration rate (W_{inf}) for its role in soil infiltration rate variation.

To investigate the role of the cover and spatial pattern of installed resource sinks in the recovery of a degraded dryland, we modified the original model to include resource-sink cells with relatively high infiltration rates, as this can be considered the most important effect of resource sinks in a water-limited system ([Ludwig and Tongway, 1996](#)). To simulate this, a fraction of the soil ($Frac_{\text{sink}}$) was initialised with a resource sink cover. These areas had a higher infiltration rate than bare soil areas in the absence of plants. Specifically, W_{inf} (eq. 1b and c), was equal to $W_{0,\text{sink}} = 0.6$ in the area with resource-sink installations, corresponding to three times the infiltration in the bare soil ($W_0 = 0.2$). This was a conservative estimate compared to the up to ten-fold increase of the soil infiltration rate measured by [Tongway and Ludwig \(1996\)](#) for resource sinks made out of branches. In all cases, as for the original model, the infiltration rate under plants was larger than these baseline values, and it increased saturating towards a maximum for large biomass (eq. (1b), first term on the r.h.s.).

To enable vegetation to colonise the landscape, we included a non-local seed bank term. In the original model, plant dispersal is approximated by a diffusion term. Yet, non-local seed dispersal is also a common reproduction strategy in drylands ([Pueyo et al., 2008](#)), and seed availability is not found to be limiting ([Beukes and Cowling, 2003](#); [Ludwig et al., 1994](#)). We thus included the third term on the r.h.s. of equation (1a) (P_{seed}): in a fraction of the spatial domain ($Frac_{\text{seed}}$) a small amount of biomass P_{seed} was introduced at each time step, to represent the emergence of seedlings from a seed bank; P_{seed} was zero elsewhere. The cells affected by seedling germination were chosen randomly across the landscape. Due to the inhospitality of the bare soil and the dominance of the diffusion term ([Padilla and Pugnaire, 2006](#)), the seed dispersal term mostly played a role in the beginning of the simulations, allowing new seedlings to grow in sink areas.

2.2. Analyses

We studied the effect of varying cover (from 0 to 100%) and spatial pattern (random and regular) of installed resource-sinks on the recovery of the system, measured by the average density and spatial pattern of plants at equilibrium.

In the random distribution, resource-sink cells were chosen randomly across the field ([Fig. 1A](#)). In the regular distribution, resource-sink cells were introduced following a regular geometrical structure,

Table 1

Parameter symbols, units, values and definitions. Parameter values are from [Rietkerk et al. \(2002\)](#), with the exception of P_{seed} and $W_{0,\text{sink}}$ (see explanation in the text).

Symbol (units)	value	Definition
c ($\text{g mm}^{-1} \text{m}^{-2}$)	10	The conversion of water uptake by plants to plant growth.
g_{\max} ($\text{mm g}^{-1} \text{m}^{-2} \text{d}^{-1}$)	0.05	Maximum specific water uptake
k_1 (mm)	5	Half-saturation constant of specific plant growth and water uptake
d (d^{-1})	0.25	Specific loss of plant density due to mortality
P_{seed} ($\text{g m}^{-2} \text{d}^{-1}$)	4	Non local seed dispersal
D_p ($\text{m}^2 \text{d}^{-1}$)	0.1	Plant dispersal
α (d^{-1})	0.2	Maximum infiltration rate
k_2 (g m^{-2})	5	Saturation constant of water infiltration
W_{inf} (-)	-	Water infiltration rate of the soil in the absence of plants; it can be equal to: W_0 (-) or $W_{0,\text{sink}}(-)$
W_0 (-)	0.2	Bare soil infiltration
$W_{0,\text{sink}}$ (-)	0.6	Resource sink infiltration
r_w (d^{-1})	0.2	Specific soil water loss due to evaporation and drainage
D_w ($\text{m}^2 \text{d}^{-1}$)	0.1	Diffusion coefficient for soil water
R (mm d^{-1})	-	Rainfall
D_o ($\text{m}^2 \text{d}^{-1}$)	100	Diffusion coefficient for surface water

aiming at recreating regular patterns (Fig. 1D, G, J). The regular distribution was created by choosing an arrangement where the distance between neighbouring patches were equal across the lattice in the x and y directions from the core of the patch, while the patch size would vary between runs (Fig. 1D, G, J). For patch distances, we repeated the analysis in a range similar to what observed in the original model for the distances between plant patches (Rietkerk et al., 2002). The exact value

of the distance between resource-sink patches proved not to be of major influence on plant density at equilibria (see Appendix A). To simulate a range of resource-sink covers between 0 and 100%, the regular distribution increased the size of resource-sink patches from single-cell up to square patches of $N \times N$ cells (with N being an odd number between 3 and 25). To verify that our results did not depend on the specific geometry chosen for the regular distribution, we repeated the simulations also

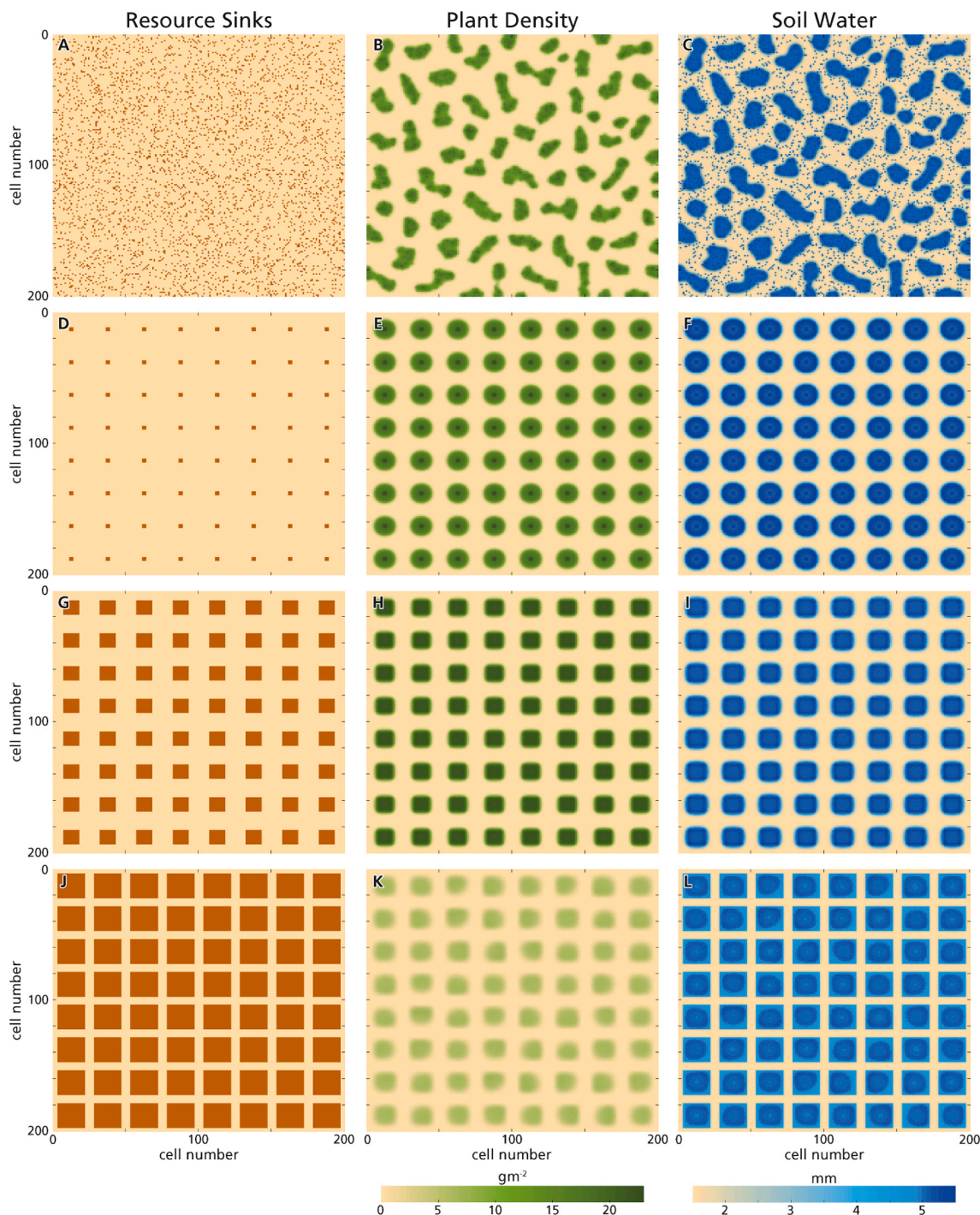


Fig. 1. Different spatial distribution and cover of resource sinks lead to different final vegetation and soil water patterns. A, D, G, J, spatial distribution of resource sinks (brown), within a bare soil matrix, following: A, random distribution ($Frac_{sink} = 10\%$); D, G, J, regular distributions. With patch sizes of 3×3 cells ($Frac_{sink} = 1.5\%$) (D), 11×11 cells, ($Frac_{sink} = 20\%$) (G) and 21×21 cells patches, ($Frac_{sink} = 60\%$) (J). B, E, H, K, the plant density at the end of the simulation. C, F, I, L, the soil water patterns at the end of the simulation. Both B, E, H, K and C, F, I, L for the corresponding resource sink distribution shown on the left in each row. See colorbars for scale: for plant density, yellow represents bare soil and green the maximum density ($g\ m^{-2}$); for soil moisture (mm) blue represents wetter soil and yellow dry soil. In all simulations $R = 0.75\ mm\ d^{-1}$; see Table 1 for other parameter values. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

with a second regular distribution of resource-sinks, where the size of the resource sink patches was kept constant, chosen to be close to the average vegetation patch size in the original model. To increase the cover of resource sinks, the number of resource-sink patches was increased between simulations (see Appendix A).

We also performed a sensitivity analysis for a few different values of $W_{0,\text{sink}}$ (Appendix B) to verify the influence of the exact value of this parameter on the results.

We run the simulations for a time period of 9000 days (approximately 25 years), with a time step of 0.005 days. Such long runs enabled vegetation to reach a stable state (Appendix C). Simulations were run in Matlab R2016b. Each simulation started with a flat two-dimensional domain of 200×200 grid cells of bare soil with no vegetation. A fraction $\text{Frac}_{\text{sink}}$ of the bare soil was covered with resource sinks (with either random or regular distribution).

We repeated the analysis for three values of rainfall $R = 0.6, 0.75$ and 0.9 mm d^{-1} . We chose this range since there the original model has two stable states: bare and vegetated. The vegetated state can only be maintained or reached if at the beginning of the simulations spikes of high plant density are introduced, i.e. for healthy ecosystem conditions, while if the model starts off with low homogeneous plant density (representing degraded conditions) all the vegetation dies out and bare soil is the final outcome of the model (Rietkerk et al., 2002). The latter state is alternatively stable to the original vegetation pattern state, or, in other words, hysteresis locks the system into this degraded state. This range of rainfall is thus appropriate to investigate whether the application of resource sinks can lead to a recovery of the system from the degraded to the vegetated state.

3. Results

In our simulations, the addition of resource sinks in a bare landscape, spatially distributed either randomly or regularly, could trigger the recovery of the system to a vegetated landscape (Figs. 1–2, Appendix A). Technically, the resource sinks destabilized the bare landscape into an alternative stable state with vegetation growing in spatial patterns, while vegetation would not grow without the presence of resource sinks for the rainfall values analysed (Fig. 2, a resource sink cover of 0%

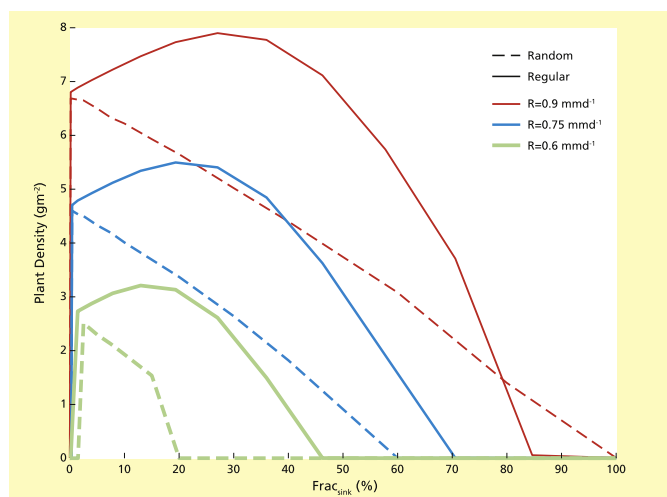


Fig. 2. Average plant density at equilibrium obtained for different sink cover ($\text{Frac}_{\text{sink}}$, %) and rainfall (R , mm d^{-1}). The almost vertical lines close to the Y-axis represent the shift from bare soil to a vegetated state, already observable for sink cover values $< 0.5\%$. Note that for $\text{Frac}_{\text{sink}} = 0\%$, plant density is close to zero. The dashed lines represent random distribution, solid lines regular distribution. The colour of the lines indicates the average annual rainfall; red, $R = 0.9 \text{ mm d}^{-1}$; blue, $R = 0.75 \text{ mm d}^{-1}$; green, $R = 0.6 \text{ mm d}^{-1}$. See Table 1 for other parameter values. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

means in all cases a plant density close to zero). Thus, not surprisingly, our model displayed a similar bi-stability of vegetated and bare soil states found in the original model (Rietkerk et al., 2002).

A small cover of installed resource sinks was sufficient to promote the recovery of the vegetation. However, the system did not recover when large covers of resource sinks were installed (Figs. 1–2). Larger applied resource sink cover generally led to a decrease in biomass, reaching zero for resource-sink cover above 50–75% (with the exact value depending on the rainfall level). The specific characteristics of this decline varied depending on the spatial distribution of the resource sinks. From the minimum cover of resource sinks that promote the recovery of the system, plant density decreases more or less linearly with resource-sink cover for random distributions, while it follows a unimodal response for regular distributions, peaking at intermediate values of resource-sink cover (Fig. 2; see also Appendix A).

For a random distribution small (single-cell) resource sinks were scattered across the entire domain (Fig. 1A), which generally prevented the existence of large sink patches. The vegetation self-organized in a spatial pattern (Fig. 1B), which started from the expansion of a few of the resource sink cells with favourable high soil water conditions across the lattice (Fig. 1C). The final vegetation pattern resembled the regular vegetation patterns observed with the original model (Rietkerk et al., 2002), yet they were less symmetric, seemingly because vegetation tended to conglomerate around as many sink cells as possible. Some of the applied sink cells did not develop vegetation, but actively subtracted water from vegetated patches, as displayed by their high soil moisture content (Fig. 1C). The number of applied sink cells between the vegetated patches inherently increased with larger applied resource-sink cover, potentially explaining the almost linear decline (Fig. 2).

With regular resource sink distribution, plant density followed a hump-shaped pattern, with a maximum value for sink cover $\text{Frac}_{\text{sink}}$ of approximately 27% ($R = 0.9 \text{ mm d}^{-1}$), 19% ($R = 0.75 \text{ mm d}^{-1}$) and 13% ($R = 0.6 \text{ mm d}^{-1}$). For larger $\text{Frac}_{\text{sink}}$, a sharp decline in plant density was observed, reaching zero around 50%–75% (with the specific value increasing with rainfall R). Vegetation developed completely over the resource sink area (Fig. 1E, H). However, if the resource sink areas were large, the sinks stopped supporting vegetation at the edges of the patch (Fig. 1K). These edges would end up subtracting part of the scarcely available water (Fig. 1L), leading to the observed plant density decline with $\text{Frac}_{\text{sink}}$ (Fig. 2). Small $\text{Frac}_{\text{sink}}$ values could support a relatively high amount of plant density. Circular vegetation patches developed around the resource-sink cells, which would support higher plant density than the rest of the vegetation patch. Noticeably, vegetation triggered by small sink patches (Fig. 1D) developed towards a vegetation patch size (Fig. 1E) similar to the $\text{Frac}_{\text{sink}}$ patches which triggered the maximum plant densities (Fig. 1G and H).

Regularly distributed resource-sink applications generated in all cases a higher biomass than the random distribution, for equal applied area (Fig. 2). This was a direct consequence of the higher water losses by evaporation and drainage in the random distribution, as more resource-sink areas remained outside the vegetation patches and the water infiltrated therein was out of reach for plants (Fig. 1C). The results presented here held within the sensitivity analyses performed for different values of $W_{0,\text{sink}}$ (Appendix B) and for different regular spatial distributions (Appendix A).

4. Discussion

In agreement with field-work studies, our model results confirm that the installation of resource sinks can restore degraded drylands that are not able to recover naturally. More importantly, our results show the importance for the success of the restoration of both the cover and the spatial organization of resource sinks (i.e., piles of woody debris, sometimes accompanied with seeds), two factors that were not investigated in concert before. Regarding cover, we found that a very small cover of resource sinks was sufficient to trigger the recovery of

vegetation. However, a high cover of resource sinks could lead to a complete failure of vegetation recovery. While the former results were observed for both random and regular spatial distributions of resource sinks, a distribution similar to that of vegetation in the reference healthy system (i.e., regular distribution in our study system) was more effective: higher plant densities were reached for a given initial cover of resource sinks.

As a recent modelling work describes, the installation of resource sinks in a restoration action can facilitate the recovery of a degraded dryland by breaking the positive feedback between resource loss and vegetation loss and, at the same time, intensify the negative feedback between resource concentration and vegetation growth (Mayor et al., 2019). Indeed, the displacement of surface water towards a given resource sink (i.e., installed resource sinks and/or vegetation patches) is maximized when the sink cover is low, as there are more areas with low infiltration (i.e., bare soil) leading to a higher net displacement of surface water to the existing sinks (Mayor et al., 2019; Urgeghe et al., 2010; Urgeghe and Bautista, 2015). This maximized source-to-sink transfer of surface water strongly enhances plant growth, and probably explains why our system was able to recover when resource sinks were installed only in a small part of the space. We estimated the cover of resource sinks used in field work experiments whenever that was possible and found out that, in agreement with our results, a very small cover (e.g., <1% of erosion barriers in Kimiti et al., 2017 and Cavallero et al., 2019) could indeed trigger the restoration of the ecosystem under field conditions.

While a low cover of resource sinks suffices to recover vegetation in a degraded ecosystem, our model results showed that high covers could be counterproductive and even lead to the complete failure of vegetation recovery. This is again related to the key role of vegetation patchiness, and its associated redistribution of surface water from runoff sources to runoff sinks, in the sustenance of dryland vegetation, which was already pointed out by the seminal work of Noy-Meir (1973) and later showed by modelling works (Aguiar and Sala, 1999; Boer and Puigdefábregas, 2005; Rietkerk et al., 2002). These models show that plant productivity decreases or even collapses if the plant density of a given patchy dryland is distributed homogeneously in space, as in this way surface runoff is not produced. Installing high covers of resource sinks in our model had a similar impact than distributing plant density more homogeneously in space: the net displacement of surface water to vegetation patches decreased, and when a certain cover of resource sinks was reached there is not enough concentration of surface water anywhere that allowed plants to establish at that level of aridity.

Unimodal responses of plant density to variation in resource-sink cover were observed for both random and regular distributions of resource sinks, however, regular distributions generally allowed higher plant densities than random distributions for a given resource-sink cover. This is explained by the tendency of vegetation in our system to form regular patches, as observed in a significant proportion of patchy drylands worldwide (Berdugo et al., 2019; Deblauwe et al., 2008). Thus, when resource sinks were applied randomly and regular vegetation patterns emerged, some of the (installed) resource sinks were left in the areas between vegetation patches, where water infiltrated at a higher rate than in bare soils, lowering the net displacement of surface water towards vegetation patches, and ultimately leading to decreased plant density.

Interestingly, for regular distributions of resource sinks in our model simulations, despite equilibrium plant density decreased when the cover of resource sinks increased beyond the optimal, equilibrium plant cover remained similar, and at a similar value than the optimal cover of resource sinks (i.e., the cover of resource sinks that allowed the highest plant density, $\approx 20\%$ in Fig. 2). Furthermore, this plant cover value was in the same range as the one in the reference healthy system of the original model when using the same parameterization (Rietkerk et al., 2002). These results are in line with previous works suggesting the existence of an optimal vegetation cover in patchy drylands for which the

capture and use of water resources is maximized (Aguiar and Sala, 1999; Ludwig et al., 1994; Puigdefábregas et al., 1999; Urgeghe et al., 2010). Of course, this optimal cover value would depend on the dryland biome and its level of aridity (Berdugo et al., 2017; Ludwig et al., 1994). Nevertheless, our modelling results suggest that it should not be exceeded when installing resource sinks to restore a degraded area, in order to achieve an optimal cost-effective action.

One of the simplifications in our model was that resource sinks did not degrade. In reality, the durability of these structures largely depends on the materials used to build them, from a relatively fast degradation of organic materials (Austin and Vivanco, 2006) to practically no degradation when stones are used (Li, 2003). Field experiments testing different types of resource sinks with biodegradable materials found that robustly-built sink structures remained intact after three years despite trampling from domestic or wild animals (Kimiti et al., 2017; Ludwig and Tongway, 1996; Mando and Stroosnijder, 1999). Three years was also the time frame for plant establishment in our model (Appendix C), similarly to field observations (e.g., Kimiti et al., 2017). In this way, even if resource sinks decayed in our model, plants would still establish and start to form patterns from the initial resource sinks cover, and thus the hump-backed response of plant density to variation in resource-sink cover would remain. However, the hump of this response curve might become wider in time, as the degradation of the resource sinks would remove the differences in equilibrium plant density for a larger range of resource-sink cover. Another simplification of our modelling study is that it represented a flat landscape. In this way, runoff was displaced from bare soils towards vegetation patches and (installed) resource sinks, and did not run off from the landscape. In sloped terrains, runoff pathways are aligned with the slope and runoff losses from the landscape increase with the spatial connectivity of the bare-soil areas (e.g., Mayor et al., 2008; Saco and Moreno-de las Heras, 2013), particularly when high runoff velocities can be expected (Boer and Puigdefábregas, 2005). In this situation, a staggered regular distribution of resource sinks would reduce runoff losses with respect to the square regular distribution used in our work, preventing runoff pathways of the length of the slope.

Our modelling work gave new applied information about the spatial design of a simple low-cost restoration action for drylands: the installation of resource sinks or resource barriers in severely degraded dryland landscapes. Several field experiments have proved the efficiency of this restoration action in drylands worldwide, and have provided useful information on the efficacy of different materials that can be used to build these structures. Our work added that the most effective cover and spatial arrangement of resource sinks were those that were characteristics of the reference healthy state, but most importantly, that a very low cover of resource sinks can already trigger the recovery of the system. Furthermore, if reference plant cover is unknown, our results showed that it is safer to choose lower than higher covers of installed resource sinks. Using too-high covers of resource sinks may disrupt the necessary redistribution of surface water towards vegetation patches that allows dryland vegetation to survive aridity, and thus compromise the success of the restoration. Even though the installation of resource sinks can be a relatively low-cost restoration measure (Kimiti et al., 2017), it is still labor intensive when applied in large areas (Beukes and Cowling, 2003). Given the high efficiency of low covers of resource sinks suggested by our work, combined with the low-cost materials needed, the installation of resource sinks in severely degraded drylands has the potential to be a key contributor to the large restoration efforts needed to achieve land-degradation neutrality in the coming decades, particularly in developing countries.

CRedit authorship contribution statement

Paul M.J. Berghuis: Conceptualization, Investigation, Methodology, Formal analysis, Software, Writing - original draft, Writing - review & editing. **Ángeles G. Mayor:** Investigation, Writing - original draft,

Writing - review & editing. **Max Rietkerk**: Conceptualization, Investigation, Writing - review & editing. **Mara Baudena**: Investigation, Methodology, Software, Writing - review & editing.

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.

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Appendices. Supplementary data

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References

- Aguiar, M.R., Sala, O.E., 1999. Patch structure, dynamics and implications for the functioning of arid ecosystems. *Trends Ecol. Evol.* [https://doi.org/10.1016/S0169-5347\(99\)01612-2](https://doi.org/10.1016/S0169-5347(99)01612-2).
- Austin, A.T., Vivanco, L., 2006. Plant litter decomposition in a semi-arid ecosystem controlled by photodegradation. *Nature* 442, 555–558. <https://doi.org/10.1038/nature05038>.
- Bathiany, S., Dakos, V., Scheffer, M., Lenton, T.M., 2018. Climate models predict increasing temperature variability in poor countries. *Sci. Adv.* 4 <https://doi.org/10.1126/sciadv.aar5809>.
- Bautista, S., Mayor, Á.G., Bourakhouadar, J., Bellot, J., 2007. Plant spatial pattern predicts hillslope runoff and erosion in a semiarid mediterranean landscape. *Ecosystems* 10, 987–998. <https://doi.org/10.1007/s10021-007-9074-3>.
- Berdugo, M., Kéfi, S., Soliveres, S., Maestre, F.T., 2017. Plant spatial patterns identify alternative ecosystem multifunctionality states in global drylands. *Nat. Ecol. Evol.* 1 <https://doi.org/10.1038/s41559-016-0003>.
- Berdugo, M., Soliveres, S., Kéfi, S., Maestre, F.T., 2019. The interplay between facilitation and habitat type drives spatial vegetation patterns in global drylands. *Ecography* 42, 755–767. <https://doi.org/10.1111/ecog.03795>.
- Beukes, P.C., Cowling, R.M., 2003. Evaluation of restoration techniques for the succulent karoo, South Africa. *Restor. Ecol.* 11, 308–316. <https://doi.org/10.1046/j.1526-100X.2003.00192.x>.
- Boer, M., Puigdefábregas, J., 2005. Effects of spatially structured vegetation patterns on hillslope erosion in a semiarid Mediterranean environment: a simulation study. *Earth Surf. Process. Landforms* 30, 149–167. <https://doi.org/10.1002/esp.1180>.
- Borgogno, F., D'Odorico, P., Laio, F., Ridolfi, L., 2009. Mathematical models of vegetation pattern formation in ecohydrology. *Rev. Geophys.* <https://doi.org/10.1029/2007RG000256>.
- Cavallero, L., Ledesma, M., López, D.R., Carranza, C.A., 2019. Retention and redistribution of biological legacies generate resource sinks in silvopastoral systems of Arid Chaco forests. *Ecol. Process.* 8 <https://doi.org/10.1186/s13717-019-0180-x>.
- Deblauwe, V., Barbier, N., Coueron, P., Lejeune, O., Bogaert, J., 2008. The global biogeography of semi-arid periodic vegetation patterns. *Global Ecol. Biogeogr.* 17, 715–723. <https://doi.org/10.1111/j.1466-8238.2008.00413.x>.
- Gilad, E., von Hardenberg, J., Provenzale, A., Shachak, M., Meron, E., 2004. Ecosystem engineers: from pattern formation to habitat creation. *Phys. Rev. Lett.* 93, 98105.
- James, J.J., Carrick, P.J., 2016. Toward quantitative dryland restoration models. *Restor. Ecol.* 24, S85–S90. <https://doi.org/10.1111/rec.12393>.
- Kéfi, S., Rietkerk, M., Alados, C.L., Pueyo, Y., Papanastasis, V.P., ElAich, A., De Ruiter, P. C., 2007. Spatial vegetation patterns and imminent desertification in Mediterranean arid ecosystems. *Nature* 449, 213–217. <https://doi.org/10.1038/nature06111>.
- Kimitt, D.W., Riginos, C., Belnap, J., 2017. Low-cost grass restoration using erosion barriers in a degraded African rangeland. *Restor. Ecol.* 25, 376–384. <https://doi.org/10.1111/rec.12426>.
- Li, X.-Y., 2003. Gravel–sand mulch for soil and water conservation in the semiarid loess region of northwest China. *CATENA* 52, 105–127. [https://doi.org/10.1016/S0341-8162\(02\)00181-9](https://doi.org/10.1016/S0341-8162(02)00181-9).
- Ludwig, J., Tongway, D.J., Marsden, S.G., 1994. A flow-filter model for simulating the conservation of limited resources in spatially heterogeneous, semi-arid landscapes. *Pac. Conserv. Biol.* 1, 209. <https://doi.org/10.1071/pc940209>.
- Ludwig, J.A., Tongway, D.J., 1996. Rehabilitation of semiarid landscapes in Australia. II. Restoring vegetation patches. *Restor. Ecol.* 4, 398–406. <https://doi.org/10.1111/j.1526-100X.1996.tb00192.x>.
- Mando, A., Stroosnijder, L., 1999. The biological and physical role of mulch in the rehabilitation of crusted soil in the Sahel. *Soil Use Manag.* 15, 123–127. <https://doi.org/10.1111/j.1475-2743.1999.tb00075.x>.
- Mayor, Á.G., Bautista, S., Rodríguez, F., Kéfi, S., 2019. Connectivity-mediated ecohydrological feedbacks and regime shifts in drylands. *Ecosystems* 22, 1497–1511. <https://doi.org/10.1007/s10021-019-00366-w>.
- Mayor, Á.G., Bautista, S., Small, E.E., Dixon, M., Bellot, J., 2008. Measurement of the connectivity of runoff source areas as determined by vegetation pattern and topography: a tool for assessing potential water and soil losses in drylands. *Water Resour. Res.* 44 <https://doi.org/10.1029/2007WR006367>.
- Mayor, Á.G., Kéfi, S., Bautista, S., Rodríguez, F., Cartení, F., Rietkerk, M., 2013. Feedbacks between vegetation pattern and resource loss dramatically decrease ecosystem resilience and restoration potential in a simple dryland model. *Landscape Ecol.* 28, 931–942. <https://doi.org/10.1007/s10980-013-9870-4>.
- McCallum, K.P., Lowe, A.J., Breed, M.F., Paton, D.C., 2018. Spatially designed revegetation—why the spatial arrangement of plants should be as important to revegetation as they are to natural systems. *Restor. Ecol.* 26, 446–455. <https://doi.org/10.1111/rec.12690>.
- Meron, E., 2012. Pattern-formation approach to modelling spatially extended ecosystems. *Ecol. Model.* 234, 70–82. <https://doi.org/10.1016/j.ecolmodel.2011.05.035>.
- Noy-Meir, I., 1973. Desert ecosystems: environment and producers. *Annu. Rev. Ecol. Systemat.* 4, 25–51. <https://doi.org/10.1146/annurev.es.04.110173.000325>.
- Padilla, F.M., Pugnaire, F.I., 2006. The role of nurse plants in the restoration of degraded environments. *Front. Ecol. Environ.* 4, 196–202. [https://doi.org/10.1890/1540-9295\(2006\)004\[0196:TRONPI\]2.0.CO;2](https://doi.org/10.1890/1540-9295(2006)004[0196:TRONPI]2.0.CO;2).
- Prävälje, R., 2016. Drylands extent and environmental issues. A global approach. *Earth Sci. Rev.* 161, 259–278. <https://doi.org/10.1016/J.EARSCIREV.2016.08.003>.
- Pueyo, Y., Kéfi, S., Alados, C.L., Rietkerk, M., 2008. Dispersal strategies and spatial organization of vegetation in arid ecosystems. *Oikos* 117, 1522–1532. <https://doi.org/10.1111/j.0030-1299.2008.16735.x>.
- Puigdefábregas, J., Sole, A., Gutierrez, L., Del Barrio, G., Boer, M., 1999. Scales and processes of water and sediment redistribution in drylands: results from the Rambla Honda field site in Southeast Spain. *Earth Sci. Rev.* 48, 39–70. [https://doi.org/10.1016/S0012-8252\(99\)00046-X](https://doi.org/10.1016/S0012-8252(99)00046-X).
- Rietkerk, M., Boerlijst, M.C., van Langevelde, F., Hillerislambers, R., van de Koppel, J., Kumar, L., Prins, H.H.T., de Roos, A.M., 2002. Self-organization of vegetation in arid ecosystems. *Am. Nat.* 160, 524–530. <https://doi.org/10.1086/342078>.
- Rietkerk, M., van de Koppel, J., 1997. Alternate stable states and threshold effects in semi-arid grazing systems. *Oikos* 79, 69. <https://doi.org/10.2307/3546091>.
- Saco, P.M., Moreno de las Heras, M., 2013. Ecogeomorphic coevolution of semiarid hillslopes: emergence of banded and striped vegetation patterns through interaction of biotic and abiotic processes. *Water Resour. Res.* 49 (1), 115–126.
- Suding, K.N., Gross, K.L., Houseman, G.R., 2004. Alternative states and positive feedbacks in restoration ecology. *Trends Ecol. Evol.* <https://doi.org/10.1016/j.tree.2003.10.005>.
- Tongway, D., Ludwig, J., 2011. *Restoring Disturbed Landscapes: Putting Principles into Practice*. Island Press.
- Tongway, D.J., Ludwig, J.A., 1996. Rehabilitation of semiarid landscapes in Australia. I. Restoring productive soil patches. *Restor. Ecol.* 4, 388–397. <https://doi.org/10.1111/j.1526-100X.1996.tb00191.x>.
- UN-DESA, 2018. *Sustainable Development Goal 15: Progress and Prospects. An Expert Group Meeting in Preparation for HLPF 2018: Transformation towards Sustainable and Resilient Societies. Background Notes for Discussion Sessions; UN-DESA Division for Sustainable Development Goals, New York, NY, USA [Google Scholar]*.
- Urgehe, A.M., Bautista, S., 2015. Size and connectivity of upslope runoff-source areas modulate the performance of woody plants in Mediterranean drylands. *Ecohydrology* 8, 1292–1303. <https://doi.org/10.1002/eco.1582>.
- Urgehe, A.M., Breshears, D.D., Martens, S.N., Beeson, P.C., 2010. Redistribution of runoff among vegetation patch types: on ecohydrological optimality of herbaceous capture of run-on. *Rangel. Ecol. Manag.* 63, 497–504. <https://doi.org/10.2111/REM-D-09-00185.1>.
- Ye, J., Delgado-Baquero, M., Soliveres, S., Maestre, F.T., 2019. Multifunctionality debt in global drylands linked to past biome and climate. *Global Change Biol.* 25, 2152–2161. <https://doi.org/10.1111/gcb.14631>.