Land degradation means a loss of management options

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Highlights

- Land degradation is examined in the context of Coupled Human and Natural Systems
- Humans use exergy in their environment, causing ecological degradation
- Land degradation is the terminal state of an ecological degradation trajectory
- Large interconversion potential between land uses can decrease land degradation
- A realistic goal would be to regulate, rather than eliminate, land degradation

Abstract:

This essay approaches land degradation by targeting its ultimate thermodynamic causes, rather than its immediate environmental consequences. The objective is to make some propositions that could help understand the essence of the process, and contribute to a theoretical framework to be developed. These propositions are: 1. Human populations are an ecosystem component, not an external driver. 2. Coupled Human and Natural Systems (CHANS) tend to increase their overall complexity over time. CHANS complexity cannot feasibly be managed. 3. CHANS are made up of two types of subsystems, a consuming Foreland (FL) consisting of the human population, and a producing Backland (BL) in its environment. 4. The FL maintains its order at the expense of simplifying the BL, which becomes an entropy sink. This is the essence of ecological degradation, which is inherent to CHANS persistence. 5. Land degradation is an ecological state, not a landscape type. Hence it should be assessed within a complete range of states of ecological maturity. 6. Land use creates degradation proportional to the simplification of the ecosystems involved. Such degradation can be defined as a decrease in exergy, and results in loss of management options. Three associated corollaries are: a) A more effective target may be to regulate rather than attempt to eliminate land degradation; b) Monitoring ecological degradation trajectories may be more effective than assessing land degradation states; c) Land degradation can be decreased by maximizing the potential for interconversion between land uses.

Keywords:
Desertification; Exergy; Land Degradation Neutrality; Land Use Land Cover Change; Sustainability; Theoretical ecology

“If you know the enemy and know yourself, you need not fear the result of a hundred battles. If you know yourself but not the enemy, for every victory gained you will also suffer a defeat. If you know neither the enemy nor yourself, you will succumb in every battle.”

Sun Tzu, *The Art of War*

1. Introduction

Land degradation is widely perceived as a process whereby land becomes oversimplified and unfit for further use by human populations. Features associated with it include, but are not limited to, reduced vegetation cover, increased runoff erosion, topsoil thinning, soil salinization, aquifer depletion and loss of habitat and biodiversity. Much of the literature only reports, describes and models such effects, and their mitigation still drives current international initiatives, such as Goal 15.3 of the United Nations Sustainable Development Goals, to achieve Land Degradation Neutrality (LDN) by 2030 (Cowie et al., 2018).

Partly because of the variety of those features, and partly because all of them involve some kind of loss with respect to a reference ecosystem, land degradation is difficult to understand and theorize. It is often subject to emotional responses, with negative connotations tending to confuse empirical evidence of features that are natural in drylands, such as cracked soil or encroaching sand dunes (Prince and Podwojewski, 2020) or shifting desert boundaries (Verón et al., 2006). Such biased views combine myths with reality and further impede a true understanding of land degradation (Reenberg, 2012). As a consequence, the immediate demands of society may be implemented, worsening the problem instead of solving it, for example, indiscriminate tree planting (Mátyás and Sun, 2014).

Nevertheless, land degradation may be tackled more scientifically on a higher level of abstraction. Perhaps the two most pressing questions are whether it can be reversed, and whether it can be avoided. Concerning the first question, ecological restoration aims at the recovery of a specific damaged ecosystem with respect to its reference state. It is strongly embedded in the ecological concepts of succession and competition, and its overall purpose is to return the degraded ecosystem to its historic ecological trajectory (SER, 2004). This focus is best suited to restoring ecosystems that have been subject to a catastrophic disturbance (e.g., wildfires, volcanoes, direct human impacts, etc.) to as pristine a state as possible. While the published standards ‘…highlight the role of ecological restoration in connecting social, community, productivity, and sustainability goals’ (Gann et al., 2019), this approach struggles with historically managed ecosystems. Mediterranean drylands may serve to illustrate this point (Cortina et al., 2011). In many cases, their landscapes have evolved under human management since the Holocene, to the point that there is little certainty as to the reference ecosystem (Vallejo et al., 2012). This has been compounded by the synergistic effects of changing climate and changing vegetation since mid-Holocene (Ganopolski et al., 1998). The use of potential vegetation to set target references, as derived from the climax concept, may be misleading, because experimental data can only come from small sets of undisturbed sites not necessarily representative of the diversity of degraded ecosystems. Furthermore, degradation does not appear as a single state, but as a range of meta-states, with transitions between them not always well known (Cortina et al., 2006), further impeding establishing reference ecosystems.

Many varied solutions have been promoted worldwide in response to environmental degradation. There are large-scale restoration plans in China and Africa, for example. In the former, restoration of the Loess Plateau (Wu et al., 2020) to slow erosion has succeeded in reducing the sediment load of the Yellow River (Wang et al., 2016) and doubled vegetation cover (Chen et al., 2015). In the latter, the Great Green Wall is an ambitious project that aims at restoring 100 million ha of land for multiple environmental and social purposes, such as sequestering carbon dioxide and reducing conflict, terrorism and migration (UNCCD and Climatekos gGmbh, 2020). In Niger, over 300,000 ha have been rehabilitated, and crop yields have increased and become more stable from year to year by promoting simple water-harvesting techniques (Godfray et al., 2010). Extensive efforts have been made to mitigate land degradation in Central Asia (Cherlet et al., 2018). This is one of the examples of the WOCAT project, which contains a wide range of case studies from around the world illustrating the success of soil and water conservation initiatives (Liniger and Critchley, 2007). In addition, it should be noted that an increasing proportion of land is under some form of protection (DeFries et al., 2004), albeit such proportion exceeds 50% only for 12% of the world terrestrial ecoregions (Dinerstein et al., 2017).

Nevertheless, mitigation approaches to ecological restoration such as those presented above focus on the effects of land degradation, and therefore, tend to be reactive and often case-specific. The contribution to knowledge of land degradation built up in this way is thus heterogeneous, and whilst efficient in providing short or mid-term solutions, does not lend itself well to process theory.

Land degradation as addressed here is associated with human over-exploitation of natural resources, which leads to the second question above, whether it can be avoided. Efforts have been made in recent years to generalize desertification (i.e., the socio-economic cause of land degradation in drylands) and land degradation in appropriate scientific terms. Prominent global examples are the Desertification Synthesis of the Millennium Ecosystem Assessment (Adeel et al., 2005), the IPBES Assessment on Land Degradation (IPBES, 2018) and the World Atlas of Desertification (Cherlet et al., 2018). While comprehensive in the description of case studies, and suggestive in identifying convergence, those initiatives still focus on how land degradation proceeds rather than on why it occurs in the first place. The question of whether land degradation is an avoidable outcome or an inherent property of the human exploitation of landscapes and their natural resources therefore remains open.

In this short essay, we approach land degradation from a neutral point of view, targeting its ultimate causes, rather than at its immediate consequences. The objective is to make some propositions that could help understand the essence of the process and, perhaps, contribute to a theoretical framework yet to be developed. From this viewpoint, the restoration, management and desertification approaches described above would be complemented with a perspective of land degradation seen from higher abstraction, not as an alternative to them. In so doing, humans are considered part of the ecosystems concerned rather than their drivers. This implies that humans and their environment are coupled subsystems that exchange energy, matter and information, within them and with their surroundings, following to some extent the approach of Coupled Human and Natural Systems (CHANS) (Alberti et al., 2011).

We address a significant question implicit in land use: To what extent should efforts be invested in managing, rather than eliminating, land degradation. For this purpose, we use, as a baseline, the United Nations Convention to Combat Desertification (UNCCD) definition: *land degradation is a reduction or loss in arid, semi-arid and dry sub-humid areas of biological or economic productivity and complexity*.

2. A thermodynamic interpretation

Whether as hunters, gatherers, pastoralists, farmers or foresters, humans extract reduced materials from their environment for their self-maintenance. Whilst there are dedicated engineering disciplines addressing such transfers, only thermodynamics enable the unifying and macroscopic perspective necessary to understand the role of land degradation in human persistence.

Adapting the principles of Jørgensen and Fath (2004), CHANS are open systems, as they have a throughflow of matter coupled to another of energy. Mass and energy are conserved. Processes occurring within CHANS require energy, which enters the system in low-entropy form (solar radiation) and dissipates into the surroundings as high-entropy energy (heat). Therefore, according to the Second Law of Thermodynamics, such processes are irreversible.

Exergy is the energy stored in a system available to perform work. In this context, work is the energy necessary to create ordered structures that move the system away from thermodynamic equilibrium. Exergy is therefore potential energy. Sometimes it is likened to the concept of free energy, but exergy enables selection of the reference state for which work is produced. This is very important at the intersection of land use and land degradation, as discussed later.

Thermodynamic equilibrium is the final state at which the system cannot perform any work: molecules are inorganic, gradients have been equalized and exergy is zero. No different from other forms of life, CHANS use input energy to move away from thermodynamic equilibrium and maintain a state of low entropy into their surroundings. This involves storing exergy, and the permanent struggle to accumulate exergy after system maintenance is referred to as growth.

Fath et al. (2004) defined ecosystem growth as '…the quantitative increase of some measure, such as biomass or throughflow, and development is the qualitative change that occurs, such as organization or information of existing quantities.' According to these authors, ecosystems, and by extension, CHANS, grow in four stages: by incorporating low-entropy energy input in the system, by increasing their size, by increasing network complexity and by increasing information. These four pathways of growth, while apparently sequential, can operate in parallel and involve either system exergy or system power (the actual energy throughflow that is being used). Jørgensen and

Fath (2004) hypothesized that, of these growth stages, the one that moves the system furthest from thermodynamic equilibrium in the long term will be selected.

The outline above was primarily designed for ecosystems in general, and for that reason it also applies to CHANS, which are hierarchical systems that can be defined on several organizational levels. Therefore, it may be helpful to examine the types of CHANS subsystems from a functional perspective.

’CHANS’ here refers not only to a target human population, but also the spatial domain where it sources matter and thrives. From a thermodynamic perspective, CHANS are made up of two types of coupled subsystems that respectively decompose and cycle waste, and fix energy (Fig. 1). This is equivalent to any other ecosystem analysis, but the particular features of humans as a social species imprint strong effects on their environment that can be detected at multiple spatial and temporal scales. Therefore, we propose that, at a landscape level, such subsystems can be identified as a Foreland (FL) and Backland (BL), respectively. Formally, the FL is the heterotrophic subsystem that contains the human population, the BL is the (at least partially) autotrophic and simplified subsystem acting as coupled environment for the FL, and the coupling between them consists of a neat transference of entropy from the FL to the BL. A third type of subsystem refers to the space CHANS are embedded in, the Hinterland (HL). Whilst not directly managed by humans, according to those definitions, the HL is the environment surrounding CHANS and therefore the ultimate thermodynamic sink, and at the same time it acts as a spatial buffer for BL to be taken from or abandoned in. It is also the ecological reference for the BL.

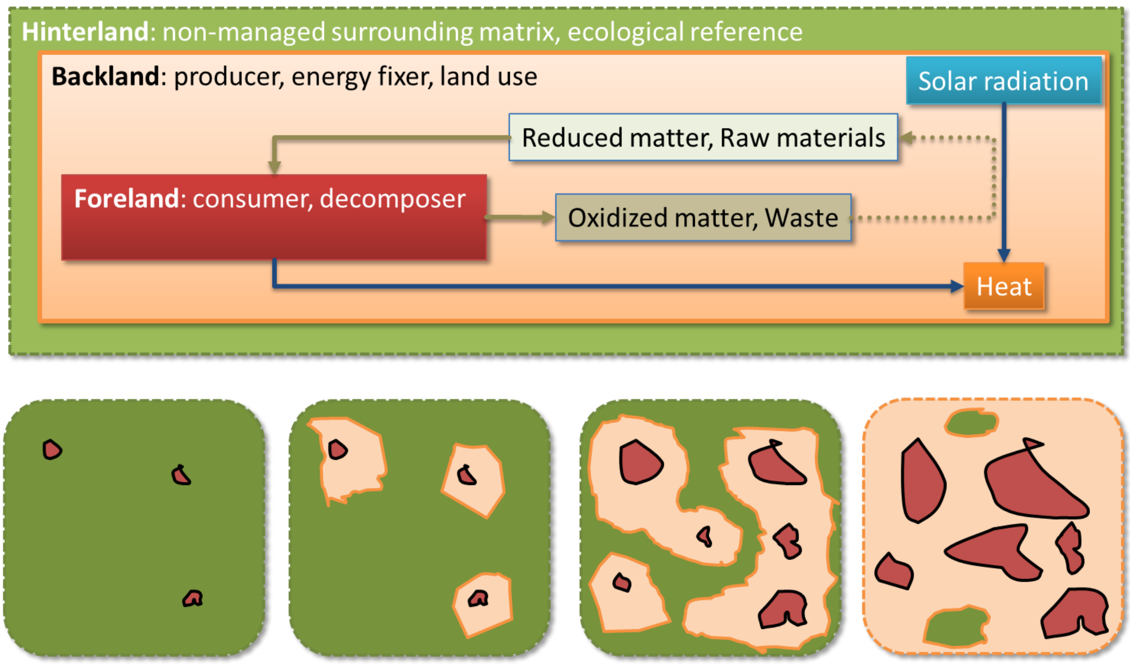

Figure 1. Coupled subsystems in a CHANS: functional interactions (above) and ideal early spatial structure dynamics (below).

These functions are not very different from conventional ecology subsystems. Palaeolithic hunter-gatherers as FL were in fact predators or super-predators who obtained low-entropy chemical energy directly from a diffuse and unmanaged HL. The advent of agriculture involved the development of a managed dedicated BL. This replicated itself as a complete ecosystem, with energy fixers (cultivated plants), consumers (farm animals) and predators (humans). From that perspective, agriculture consists of creating a simplified ecosystem where energy-fixing (i.e., primary productivity) is maximized at the expense of structure-creating functions (i.e., biomass).

Taking unintervened ecosystems as a reference, this local loss of complexity is, by definition, a first step toward ecological degradation. From this point of view, ecological degradation could be defined as the use of exergy, and its associated increment of entropy, in the coupled surroundings (the BL) of a growing human population (the FL). The BL is therefore the sink for the entropy produced in the FL, and ecological degradation in the CHANS thus becomes inherent to human persistence.

Put that way, any heterotroph could be attributed ecological degradation just for self-organizing coupled to its environment, which obviously is not the case. This is an important point that must be emphasized. Humans are basically like any other heterotroph. The difference lies in the multi-scale spatial and temporal impact they have had on their environment, especially since the start of the Neolithic. This impact arises from highly efficient modification of the environment for food production, and is what justifies using the concept of ecological degradation, which would otherwise not be necessary to describe other species. When Neolithic humans settle in a pristine landscape, they start modifying it to create a FL and a BL. Changes are initially mild and slow, and become more severe and faster as time advances. Eventually, the BL may no longer be sustainable and reaches a degraded state, characterized by the features mentioned at the beginning. In this scheme, ecological degradation would be a gradual process in which land degradation is the terminal state.

Land degradation trajectories have been documented for several case studies. A good example is overgrazing-driven shrubland encroachment in the Chihuahua Desert, which has been described, modelled and monitored (Browning et al., 2014; Grover and Musick, 1990), and even it has a dedicated website to explore state and transition models (NRCS, 2020). Nevertheless, that type of studies has percolated little to the policy management. As a result, a frequent issue with how land degradation is conventionally addressed is that the problem is only recognized when an ecosystem has reached its terminal state. Making an analogy with the development of a disease, it would be like discarding diagnosis in favour of autopsy. Hence, a function is needed that enables the whole trajectory of a landscape under human management to be monitored along a conceptual continuum between the initial and final states (Pickup et al., 1998). Exergy and entropy are appropriate for describing that evolution, and the above definition of ecological degradation addresses human exploitation of natural resources while remaining parallel to other heterotrophs. That said, other heterotrophs could also degrade their environment if they became decoupled from it, for example, large herbivores in Africa enclosed in relatively small reserves.

It should be noted that the spatial structure shown in Fig. 1 corresponds only to the most basic stages of CHANS complexity, probably associated with an increase in size within the growth forms presented above. This would apply to a rural system with subsistence agriculture. As the system keeps growing, agriculture intensifies and starts exporting products to distant locations, thereby increasing its network complexity and overall system information, while shifting spatial scales (Nyström et al., 2019). At that point the system is likely to escape the topological constraints of mapping due to telecouplings (Díaz et al., 2019; Yu et al., 2013). In fact, we think CHANS cannot be mapped beyond a (probably low) complexity threshold, and alternative tools, such as social-ecological network analysis (Bodin et al., 2014) or System Dynamics (Ibáñez et al., 2008) would have to be used instead. Nevertheless, such approaches might be helpful in illustrating quantified transactions of energy and matter between the coupled CHANS subsystems described. Then, rather than considering each subsystem an object within its own environment, it could be defined as one of two locally bound *environs* as in Network Environ Analysis (Fath, 2012; Fath and Patten, 1999). An input *environ* would thereby receive all transactions both from other subsystems and across the outer system boundaries, and an output *environ* would send transactions to other subsystems and outside the system. This would represent CHANS flows better, and could therefore holistically explore its behaviour. For example, Fang et al. (2014) applied this method to investigate a socio-economic water system in China, using water as flow currency, and they found that only the combination of three structural and throughflow variables (cycling, indirect effects and aggradation) yielded a realistic view of the system water use efficiency. Those variables could be similarly explored in a CHANS, after setting flow currency to energy and carbon, to test its dependency on flows across the system boundaries or effects of throughflow variables on CHANS efficiency compared with natural systems. The challenge of representing land use systems through such an energy and matter network is the selection of appropriate spatial and temporal aggregation scales for the CHANS subsystems, so that the need to apply steady states does not affect network connectivity and synergism (Fath et al., 2013; Shevtsov et al., 2009).

According to the second law of thermodynamics, only a fraction of BL energy is available for transfer to the FL. In addition, part of the exergy must be invested in maintaining the BL in a simplified state to counterbalance its natural trend to grow through self-organization. As a result, the production of entropy of the whole CHANS is enormous, and the BL becomes exponentially large as the overall system complexity increases (Krausmann et al., 2013). BL performance is therefore a limiting factor for the growth of CHANS, and its exhaustion may lead to system collapse.

3. Ecological functions concerned

If the proposed thermodynamic interpretation of a CHANS is accepted, the next step is an ecological description of those coupled subsystems, which will help understand them and facilitate the selection of ecological functions appropriate for assessing land degradation as we have done in the following section.

The subsystems described above can be interpreted within the framework of ecological succession. In general, an ecosystem departing from a basic state will tend to grow and become more complex for the reasons and by the pathways explained in the section above. Its spatial footprint will increase, as will the biomass it contains, number of species and their associated trophic networks, and so will the information stored in its spatial and temporal structures and in its genes. Initially, the colonization of new space involves the production of new plant biomass (net primary production, NPP), especially as much chlorophyll-bearing tissue as possible. But, as this self-organizing trend proceeds, the progressively scarcer space and biomass already grown will cause most of the fixed energy to be spent on respiration (maximizing gross primary production). Table 1 summarizes those functions both in mature and in simplified ecosystems.

In CHANS, human exploitation of the BL by the FL keeps the BL in early, simplified stages, for which the HL is the reference. This intentional transformation, referred to as ecological degradation in the section above, involves a number of changes in ecosystem function (Table 1), but probably, the most relevant are maximizing and forcing NPP to obtain crops and meat-producing animals, and reducing 'unnecessary' biomass accordingly to minimize energy losses through respiration. This increases the turnover (i.e., the ratio of NPP to biomass) and explains, for example, why most of the biomass is periodically replaced in cropfields.

Table 1. Functions associated with opposite trends of ecosystem self-organization and human exploitation (after Margalef (1974)).

Self-organization	**Exploitation**
Nutrients captured in biota	Nutrients external to biota
Increased biomass	Low biomass
Increased gross primary production	Increased net primary production
Delayed turnover	Fast turnover
Lower density of photosynthetic pigments	High density of chlorophyll
Stable transport paths	Changing transport paths
Adaptation to fluctuations	Opportunistic adaptations
Specialized niches	Broad niches
Increased ecological diversity	Lower ecological diversity
Large patches	Smaller patches

The simplified rationale above should not lead to the false impression that any dynamic equilibrium between self-organization and exploitation is pendular, swinging back and forth according to the relative strengths of the trend. Such a model, according to Clements' (1916) Theory of Ecological Succession, should be taken as conceptually elementary and likely to apply only to fine-scale changes in the temporal dimension. At the present time, it is widely accepted that multiple stable states can coexist with relatively fast transitions between them (Scheffer et al., 2001), triggered when gradual pressure surpasses a certain threshold (Bestelmeyer et al., 2017). While such state-and-transition models are not developed further here, it is important to acknowledge the need to detect different states in BL condition to assess its evolution under opposing forces of self-organization and exploitation.

4. Assessing land condition

Land degradation always represents a stage in the evolution of some original landscape, and whilst it has its own features, in this sense it is not a landscape type in itself. Therefore, its detection and assessment should be done within a framework that covers the entire range of landscape evolution, including also ecologically mature and intermediate states. Land condition here refers to that range, where land degradation is only a particular case (del Barrio et al., 2010).

Ecosystemic approaches make up a suitable subset of methods available for assessing land degradation, because they focus on some ecosystem function changing proportional to the maturity-degradation polarity (Verón et al., 2006). These approaches use ecological functions, and are often expressed as efficiency ratios that implicitly convey their respective assumptions (Puigdefabregas et al., 2009). Among them, biomass and productivity methods use annual averages of the former, and seasonal or inter-annual peaks of the latter, respectively, as surrogates of ecosystem capacity for long-term biomass maintenance, and its resilience for recovering from disturbance. This suits the most widespread perception of land degradation and is very close to the UNCCD definition.

In theory, any ecological function could be used to assess land degradation in the above approach. In practice, however, the choice should be guided by the purpose of assessment. For example, while loss of biodiversity has been shown to be a consequence of land use and degradation (Newbold et al., 2015), using it for monitoring the process would lack anticipatory value, by simply certifying defunct species. On the contrary, functions linked to the energy flow through ecosystems, such as NPP, react faster to disturbance and enable corrective action to be taken more promptly. In addition, they are better suited to the landscape scale and have appropriate surrogates with Earth Observation techniques, which is why they are widely used in modern approaches to land degradation (Gibbs and Salmon, 2015).

Holling (1986) described the dynamic behaviour of ecosystems as the sequential interaction of four functions (and attributes): Exploitation (early successional stages with prevalence of r-strategists, pioneer, opportunistic species), Conservation (mature, consolidated, climax-like stages with dominance of K-strategists), Creative destruction (catastrophic disturbances, senescence) and Renewal (decomposition and mineralization on the one hand, retention of nutrients on the other). Only the transition from Exploitation to Conservation is slow, and it represents the polarity between maturity and simplification discussed above. In that framework, the role of humans in the BL would be to keep it within the Exploitation phase, avoiding stored capital (exergy, biomass) accumulated by self-organization. Because a system maintained as such would rapidly become exhausted, its management shares an intentional application of the Renewal function, which may bifurcate into sustainability or degradation.

The evolution of an ecosystem under human exploitation was formalized in terms of energy flows by Pickup et al. (1994): Both annual mean biomass and NPP are expected to decrease with land degradation in the long term, but NPP peaks at its maximum at intermediate degradation states (Fig. 2). This conceptual model underlines two important requirements with methodological implications, the need to account for a complete range of land condition states and the need to distinguish between states and trends, with their respective spatial and temporal dimensions. For example, Prince (2016) implicitly incorporated it in his definition of five progressive degradation states, and del Barrio et al. (2016) explicitly explained it as rules operating with turnover and biomass to define ten land condition states.

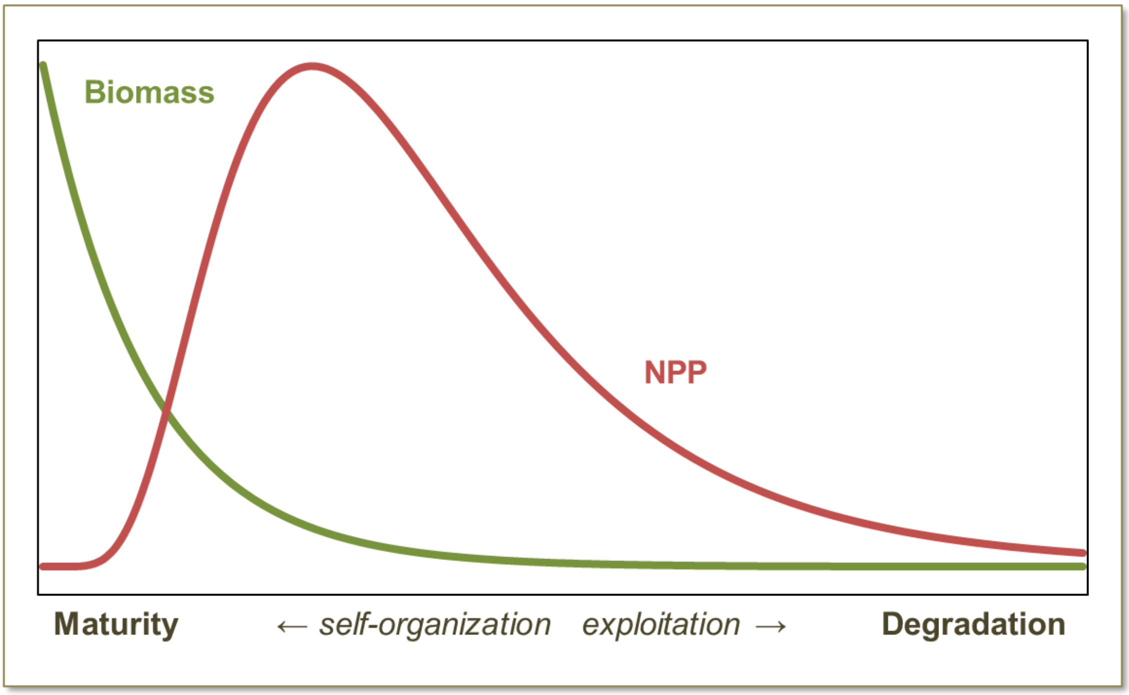

Figure 2. Changes in ecosystem biomass and Net Primary Productivity under opposing forces of self-organization and exploitation (after Pickup et al., 1994).

In drylands, NPP is strongly dependent on rainfall (P), thus Rain Use Efficiency (RUE = *NPP/P*) is preferred because it is more stable, and normalizes the amount of green biomass produced per unit rainfall. RUE was originally defined by Le Houerou (1984) to assess primary production in the steppes of northern Algeria, and Huxman et al. (2004) demonstrated its wider application by finding convergence of many different biomes at a common maximum in dry periods. Since then, RUE has been considered a reliable proxy for land condition in drylands, because high RUE scores can only be found where a complete soil structure can support vegetation functions. It is commonly implemented in large territories by using the integral of a remotely sensed vegetation index (for example, the Normalized Differences Vegetation Index, NDVI) as a proxy for aboveground NPP (Fensholt et al., 2015; Prince et al., 1998).

However, biomass throughput only provides a partial picture of actual land degradation. In addition, BL management by FL socio-economic systems and associated variations in ecosystem health, soil quality and water reserves must also be accounted for. A baseline and at least an implicit distinction between degradation states and trends must be made. Building on that, the LDN scientific framework suggests combining three global indicators: land cover change, NPP and Soil Organic Carbon (SOC), respectively, in a 'one-out, all-out' approach (Orr et al., 2017). These indicators have complementary temporal scales, and their explicit recognition represents a relevant leap forward toward an integrated notion of land degradation. This notion guides the following section.

5. Management options

It does not take much effort to accept that land use affects land condition, but, what about the opposite? Obviously, any concrete piece of land must meet some requirements for a given land use to be allocated to it, and a search for 'land suitability' for different land uses will produce a plethora of references describing precise biophysical requirements. But the question here is not which land use can be optimally allocated to a piece of land, but how many land uses may be developed there, irrespective of whether any of them is finally selected, as a function of its condition. This is potential, and we refer to it as management options.

This question may be empirically explored in a territory where both land uses and land condition are known. To operate on a realistic basis, here we discuss the results of a RUE-based assessment of land condition made in Spain for 2000-2010 (Martínez-Valderrama et al., 2016; Sanjuán et al., 2014) using land cover classes from the CORINE LC for the reference year 2006 (EEA, 2016). Appendix A describes the data preparation and statistical tests with which significant relationships between land condition and land uses were found.

That land condition assessment of Spain may be a good example of application of the approach described in the preceding section. RUE was implemented on two temporal scales to account for long- and short-term vegetation responses (del Barrio et al., 2010). A mean observed RUE was computed for every hydrological year and then averaged for the ten years of the period, yielding a proxy to maturity or biomass. In parallel, an extreme observed RUE was computed using the maximum NDVI value found in the time-series for each pixel, and its total precipitation of the six preceding months; this was taken as a proxy to resilience or productivity. Then, each of these observed RUE images was plotted against a correspondingly implemented aridity index. Each resulting scatterplot did convey the variation in the territory of the respective RUE implementation. Functions were then fitted to define the upper and lower scatterplot limits, therefore obtaining empirical references for maximum and minimum potential RUE, both along an aridity gradient and across any aridity level. The observed RUE values were then made relative to such references to enable direct comparisons between locations under different climate. Finally, each pixel was assigned a land condition state according to its position with respect to the empirical references for mean observed RUE, and to its scores of mean and extreme relative RUE being above or below the median for its aridity level (Table A1).

The concrete aspect to be explored here was whether land cover classes, which represent different land uses to some extent, can be associated with particular states of ecological maturity. In addition, if so, whether land cover classes can be grouped in land condition levels using those states.

As a result, land cover shows significant differences in land condition, so that classes with higher tree cover and less agricultural intensification relate to higher maturity states (but see below some comments in this respect). Moreover, such differences are not uniformly distributed: the eighteen land cover classes were grouped in twelve homogeneous groups. Such groups can be arranged in increasing order of ecological maturity using their typical frequency of land condition states, thus they were considered as land condition levels (Fig. 3).

LAND COVER	LAND CONDITION LEVELS											
	12	11	10	9	8	7	6	5	4	3	2	1
Coniferous forest	****											
Mixed forest	****	****										
Broad-leaved forest			****									
Permanently irrigated land			****									
Pastures		****	****	****	****	****	****					
Transitional woodland-shrub					****							
Fruit trees and berry plantations					****	****						
Agro-forestry areas					****	****						
Sclerophyllous vegetation						****	****					
Natural grasslands						****	****					
Open spaces with little or no veg.							****					
Agriculture with natural vegetation				****				****				
Non-irrigated arable land									****			
Rice fields								****	****	****		
Complex cultivation patterns										****	****	
Olive groves												****
Vineyards												****
Annual crops assoc. with perm. crops											****	****

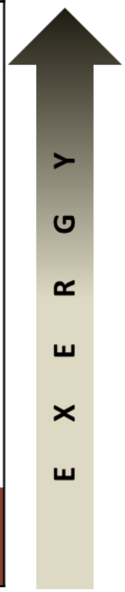

Figure 3. Land cover classes (rows) grouped by land condition level (columns). Results of a Tukey's Unequal N HSD test to detect homogeneous groups (land condition levels) of CORINE LC (2006) land cover classes, using ordinal land condition states derived from 2dRUE-ES (2000-2010). Land cover classes are in order of decreasing ecological maturity (rank sum, not shown), and membership to land condition level is marked by significance asterisks [α<0.0001, error MS = 1411E5, *df*=49029]. See Appendix A for details.

Two important comments should be made on this result before proceeding further. First, Fig. 3 looks ambiguous, as a land cover class may be included in more than one condition level. This is not an uncommon outcome of the Tukey test. When the number of observations is small, the usual interpretation is that the test could not determine which subpopulation a sample (land cover class) was drawn from. However, in this case, the dataset is large, and land cover classes may be presumed to be internally heterogeneous, as they are only a proxy to land uses. Pasture is a good example of this, both because of its broad definition within the CLC classification and its regional or even local differences in management practices. Therefore, we used the broader sense of that ambiguity to our advantage, and interpreted shared land cover classes as links between land condition levels.

Second, Fig. 3 has inconsistencies in the resulting order of land cover classes by ecological maturity. For example, it is widely accepted that broad-leaved or mixed forests have higher ecological maturity than conifer forests, and the same is true for natural grasslands with respect to pastures. This was mainly caused by the large size of the study area, which included all the drylands of mainland Spain. The classes mentioned represent landscape types rather than specific uses (e.g., rice fields or vineyards at the opposite ends of precision), and their diversity increases beyond management practices with the size of the study area. Land patches from distant locations that do not belong to the same succession scheme or land use system become merged in the same class, and are then evaluated on a common land condition scale with the only requirement being that they share similar aridity. A further inconsistency is the high position allocated to permanently irrigated land, which relates to its water input from topographic, rather than atmospheric, origin. They therefore appear as overperforming anomalies in the land condition assessment. This is somewhat coherent with the method, but should be accounted for in evaluating maturity rank because in the original study, it was used precisely to detect small land areas which are problematic because of their potential degradation. Therefore, the organization of land cover classes in Fig. 3 has certain limitations due to noise.

Overall, we considered the errors addressed in the comments above concerning the higher level of detail to be compensated by the representativeness derived from working with a large dataset from a large study area. Hence the analyses yielding Fig. 3 are still useful as long as only their coarser outcome is considered.

With that in mind, the horizontal links made by land cover classes between land condition levels in Fig. 3 are consistent with the emerging paradigm that desertification really consists of shifting land (condition) states within concrete land uses (Bestelmeyer et al., 2015). By interpreting only management differences, it follows that land condition can be improved without changing land use.

At the same time, the vertical ordination of land cover classes in Fig. 3 resembles, with the aforesaid precaution, an exergy gradient as described in Section 2 above. This does not mean that any two contiguous land cover classes may be compared precisely in terms of exergy. First, because the CLC classes are too heterogeneous to define comparable ecosystems. Forests are particularly sensitive to this, as CLC considers any land patch covered by a dense tree formation forest, irrespective of its ecological attributes. Second, because ecological exergy must be determined by quantifying organic matter and defining a chemical equilibrium (Jørgensen and Norsnielsen, 2007). Nevertheless, the top of that order tends to include land cover classes with more biomass, and biomass has been demonstrated to be proportional to ecological exergy (Jørgensen, 2002). Looking at the extremes, forests located on top of the column are obviously more ecologically grown (in terms of size, network complexity and information) than annual crops at the bottom, and the same is true for intermediate positions.

In view of the preceding arguments, another possible interpretation of Fig.3 would offer realistic clues to how to change land condition through land use interconversion. Upgrading a piece of land with a certain use and condition would involve, first, improving its condition (horizontally) up to a level that is shared (vertically) by the next higher exergy land use. This process is then repeated until the target land use is reached. The contrary is also true, and probably conveys the land degradation story of many territories: a piece of land is overused (i.e., decreasing its condition horizontally) until the current land use is untenable, then it is downgraded vertically to a next lower exergy use that accepts the decreased condition. Again, the process may be repeated down to the lowest exergy state. In both cases, the cost is higher if upgrading land condition because exergy must be stored, and lower if degrading land condition because exergy becomes spent. This defines the thermodynamic sense of land use change trajectories.

That hypothetical interpretation should be tested in a smaller study area with coherent land use systems with known interconversion stories. Short of that, we find that such an empirical step-like pattern of interaction between land use and condition suggests a plausible trajectory for change in land use. It also offers, in turn, an indirect way to improve land condition by managing land use towards maximizing their interconversion potential. Of course, it might always be possible to leapfrog across several horizontal land condition levels or even vertical land uses. However, the farther apart the initial and final state are, the more costly the operation would be. Even land degradation may be an expensive enterprise if it is not parsimonious.

In other words, more exergy means more capacity to perform work, which in turn means more potential uses to be developed, thus more management options. Considering that land uses make up the BL of the coupled systems forming CHANS, and accepting that these represent variable levels of simplification needed to keep FL entropy low, it follows that land degradation means a decrease in system exergy, and hence loss of management options, which we believe to be the critical point of this essay.

We acknowledge that the rationale explained above is speculative, and supported by a somewhat forced interpretation of some statistical facts. Nevertheless, the question remains of how land use systems meet thermodynamic laws and whether these could support a theoretical view of land degradation. If this is accepted, the challenges ahead include formalizing detected land use trajectories into appropriate state and transition models, quantifying their exergy, and formalizing their function as a BL within concrete CHANS. To this latter respect, as suggested in Section 2, energy and carbon convey the dissipative character of transactions between BL and FL subsystems. The interconversion between reduced and oxidized forms of carbon is right in the core of such transactions. Interestingly, Sustainable Development Goal 15.3 (LDN) is being monitored through Indicator 15.3.1 (Proportion of land that is degraded over the total land area) which uses three sub-indicators that have been addressed in the above discussion: trends in land cover, land productivity and carbon stocks. This may facilitate international acceptance of this approach.

6. Concluding propositions

Our purpose here was to present a system in which land degradation is consistent with an ecological analysis. We believe this or other equivalent exercise is necessary before concrete degradation problems can be tackled to avoid addressing them as local anomalies, which is often the case, rather than as a system outcome. Because of the exploratory, and admittedly conjectural, nature of this essay, its result is somewhat abstract and does not include associated experimental testing. This is why we prefer to call the following statements propositions instead of hypotheses:

1. Human populations are an ecosystem component, not an external driver. Such ecosystems, like any other, move away from thermodynamic equilibrium by maintaining local entropy sinks.

2. Coupled Human and Natural Systems (CHANS) tend to increase in overall complexity over time. This growth results from a struggle to escape from thermodynamic equilibrium. As this system behavior pertains to an organization level higher than human populations, shifting CHANS complexity cannot feasibly be managed.

3. CHANS are made up of two types of subsystems: a consuming Foreland (FL), consisting of the human population, which maintains its order at the expense of simplifying a producing Backland (BL). These are embedded in a third type, the Hinterland (HL), which remains undisturbed and serves as an ecological reference. The separation between FL and BL occurs on many temporal and spatial scales. Due to the dissipative character of the interaction between FL and BL, and considering the proposition above, a CHAN collapses if the BL is exhausted.

4. Ecological degradation is defined as the use of exergy, and its associated increment in entropy, in the coupled surroundings (the BL) of a growing human population (the FL). The BL is therefore the sink for the entropy produced in the FL, and ecological degradation in the CHANS is thereby inherent in human persistence. Land degradation occurs in the BL, and is the terminal state of a trajectory of ecological degradation.

5. Land degradation is an ecological state, not a landscape type. Hence it should be assessed within the complete range of states of ecological maturity. NPP, biomass and turnover are key variables because they describe the flow of energy and matter over the CHANS boundaries. These variables can be recorded either directly in the field or indirectly through geomatic surrogates.

6. Land use and land condition maintain interactive relationships that need to be further formalized. Land use creates degradation proportional to the simplification of the ecosystems involved. Such degradation can be defined as a decrease in exergy and results in loss of management options.

If the above are true, three corollaries arise from them:

A. A more effective target may be to regulate, rather than attempt to eliminate land degradation. This may be achieved, for example, by introducing additional cycles to slow down the production of entropy, by increasing waste recycling or by not reaching oversimplification of the BL. An intriguing question in this regard is whether modern hydroponic agriculture in greenhouses could incorporate the low-entropy energy input into the FL boundary, which would reduce the pressure on the BL.

B. Monitoring ecological degradation trajectories may be a more effective way to obtain early warnings in land management than assessing land degradation states.

C. Degradation of a territory can be decreased by maximizing the potential for interconversion of its land uses. Thus, instead of creating landscape heterogeneity by attempting to compensate heavily degraded areas with newly restored ones, it might be preferable to maintain the overall territory condition, which could be achieved through appropriate policies targeting potential land use change or rotation. Furthermore, such a policy would increase resilience of both farmers and overall system under changing conditions.

The above propositions and their associated corollaries represent the possible outcome of a logical rationale for land degradation, a subject that is often undertaken by appealing to emotions rather than facts. The approach is fresh and its formulation is likely to evolve, perhaps rendering obsolete this essay. However, it is the drive to understand land degradation that really matters, not any particular system of statements. In the words of Ludwig Wittgenstain: “My propositions serve as elucidations in the following way: he who understands me finally recognizes them as nonsensical, when he has used them as steps to climb beyond them, on them, over them. (He must, so to speak throw away the ladder, after he has climbed up it.)” (*Tractatus Logico-Philosophicus*, Proposition 6.54).

8. Appendix A

This appendix describes the datasets and statistical analyses made to explore associations between land uses and land condition described in Section 5 of this article. The dataset is open access and may be downloaded (del Barrio et al., 2020).

The land condition data were extracted from a study of Spain (2000-2010) based on a geomatic procedure that uses archived time-series from SPOT-VEGETATION S10 NDVI with a 1-km spatial resolution, and the corresponding climate fields (Sanjuán et al., 2014). The resulting land condition states describe ecological maturity in terms of NPP and biomass, and are related to the aridity found at each location. Table A1 shows their names and relevant attributes. These states were statistically validated against Soil Organic Carbon (SOC), using the map of organic carbon in topsoils in Europe (Jones et al., 2004). Three significant facts resulted: there is direct proportionality between ecological maturity represented by land condition states and SOC levels; in general, each state discriminates its own SOC level; and lower condition states trend to contain less than 2% of SOC, a threshold below which it is recommended either to change land management or to supplement organic matter. See the publication cited for full details.A summary is available in Martínez-Valderrama at al. (2016).

Table A1. Summary of land condition states based on Rain Use Efficiency (RUE) scores related to the potential minimum and maximum RUE boundaries found for a location by aridity level. States 3 through 6 fall between these boundaries, and result from further subdivision according to Net Primary Productivity (NPP) and biomass (B) being above (high) or below (low) their respective medians.

Order	Name	Parameterization
1	Underperforming anomaly	Below the minimum expected RUE confidence interval
2	Baseline performance	Within the minimum expected RUE confidence interval
3	Degraded	Low NPP, low B
4	Productive with low biomass	High NPP, low B
5	Productive with high biomass	High NPP, high B
6	Mature	Low NPP, high B
7	Reference performance	Within the maximum expected RUE confidence interval
8	Over-performing anomaly	Above the maximum expected RUE confidence interval

Land condition states were managed as ordinals in the analyses described below, with order number increasing with ecological maturity (Table A1).

Land use data for this exercise were approached using land cover classes from the CORINE LC (CLC), Level 3, for reference year 2006 (EEA, 2016). The original 250-m resolution was rescaled to 1000 m to match the land condition dataset. This was done by finding the statistical distribution of land cover classes at the 250 m resolution within each 1000-m resolution target cell, and allocating the modal class.

The study area was restricted to the 363,002 km^2 of drylands (i.e., arid, semi-arid and dry sub-humid zones according to the FAO-UNEP aridity map accompanying the land condition dataset) in mainland Spain. Of this total, only agricultural or forestry land uses (see list in Fig. 3) were entered in the statistical analysis. A stratified-random sample of raster cells meeting those requirements was drawn to ensure spatial independence of the data in the statistical tests. The 'Bare rock', 'Sparsely vegetated areas' and 'Burnt areas' land cover classes did not produce enough sample points and were merged with the higher CLC Level 2 class '*Open spaces with little or no vegetation*'. The resulting dataset had 49,047 observations of land condition (8 ordinal states) and land cover class (18 nominal classes).

The analysis was aimed at determining, first, whether there were any differences in land condition between land cover classes, and second, if such differences were detected, whether they were uniformly distributed (i.e., each land cover class was different from all others), or whether homogeneous groups of land cover classes could be formed. The significance threshold for rejecting the respective null hypotheses was set at α=0.05.

For the first aim, we applied a *Kruskal-Wallis one-way analysis of variance by ranks*. Land condition scores (Order in Table A1) of all the observations were ranked in a single series, then grouped by land cover classes, and then the sums of ranks were computed. The test enabled the null hypothesis that the average ranks found for land cover classes were drawn from the same population to be rejected, hence differences in land condition states between land cover classes were accepted (KW = 11784.46, df = 17, p < 10E-3).

After that, we only knew that at least one of the land cover classes was significantly different from at least another one in terms of land condition states. To learn more about how those differences were distributed, we made pairwise comparisons between the land cover classes, using the *Honestly Significant Differences (HSD) Tukey* test. The rank sums of the 18 land cover classes were ordered by increasing magnitude, then pairwise differences were tested following two rules: i) prioritizing pairs according to the size of their difference (i.e., largest against smallest, largest against next smallest, …, next largest to smallest, etc.), and ii) if no difference was found within a pair, subsequent pairs enclosed by that pair were not tested.

The above exercise resulted in 12 homogeneous groups of land cover classes (error MS = 1411E5, df = 49029). Fig. 3 shows the 18 land cover classes in order of decreasing rank sum (this value is not displayed), and their membership of each of the 12 homogeneous groups marked by significance asterisks. Because ranks were derived from ordinal land condition states, the 12 homogeneous groups can be interpreted as land condition levels, from 1 (lowest condition) to 12 (highest condition).

Further details on the tests applied in this analysis can be found in Zar (1984).

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