

CeNDEF Working Paper 26-02

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January 2026

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Resilience of a dry grazing system under different pastoral preferences

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Abstract

Arid and semi-arid regions around the world are inhabited by pastoralists who depend on livestock rearing for their subsistence. Exogenous shocks like droughts can cause regime shifts that threaten both ecological stability and economic livelihoods. We incorporate pastoralist economic incentives and preferences into an existing dynamic eco-hydrological grazing model exhibiting bi-stability by coupling off-take decisions to a weighted utility function, describing value generated by animal sales, ongoing benefits like milk and herd size for status and insurance for future setbacks. We find that higher prices, or equivalently, better market access for pastoralists, increases the system's resilience to irreversible shocks if pastoralists maximize their utility. On the other hand, we find that higher prices do promote management strategies with less resilience to shocks that are reversible. This means that while damage to the ecosystem can be restored, economic livelihoods will come under stress doing so, because herd size and/or off-take levels have to be reduced during this process. In addition to these findings, we present a mapping of the trade-off between utility and resilience by plotting the efficient combinations on Pareto frontiers. Higher prices steepen these frontiers, making resilience increasingly costly in terms of forgone utility. However, different abiotic conditions such as rainfall and soil type also affect these frontiers, which allows us to identify under which conditions it is attractive to trade in utility for resilience. This paper adds both to the literature about resilience in social-ecological systems in general and resilient management of rangelands prone to irreversible collapse specifically.

Keywords: Grazing systems, Resilience, Desertification, Tipping Points, Natural Resources

1. Introduction

Arid and semi-arid regions cover a substantial portion of the Earth's surface. Limited rainfall and high evapotranspiration make these drylands largely unsuitable for crop agriculture. Instead, they consist of grasslands whose inhabitants depend on livestock herding for their livelihoods. These pastoral systems are vulnerable: drought, overgrazing, and poor land management can cause grasslands to degrade into an unwanted desert state through a process known as desertification (D'Odorico et al., 2013). A key question for these systems is how economic incentives affect these state transitions.

Regime shifts in bi-stable drylands have been shown to happen over the span of decades and have been attributed to a multitude of factors. For example, Kelly and Walker (1976) reported that by 1966, about half of the Tribal Trust Lands in Rhodesia were overgrazed or had little grass cover. Enfors and Gordon (2007) demonstrated how institutional changes, deteriorating climatic conditions, and population growth caused a shift from a productive to a degraded state in Tanzania's Makanya region. Wendling et al. (2019) showed that long-term precipitation changes induced regime shifts in Northern Mali. Helldén (2008) calibrated a model over the duration of a century in Sudan's Kordofan region, showing that human pressure can drive desertification that it is not necessarily irreversible. The decline of Mongolian grasslands over the last decades has been attributed to overgrazing (Fang and Wu, 2022; Hilker et al., 2014).

In this paper, we map the trade-off between economical payouts and ecological resilience in arid and semi-arid grazing systems. Therefore, we use the eco-hydrological model introduced by Walker et al. (1981) and van de Koppel et al. (1997) to describe the positive feedback between soil water and vegetation that results in the typical bi-stability of these systems. Following van de Koppel and Rietkerk (2000), we include a dynamic population of herbivores that adapts to the available vegetation, but is additionally subject to constant harvesting by a human manager. Our model is a predator-prey model with the prey (vegetation) under critical depensation and with constant harvesting on the predator (herbivores) population. By choosing the extraction level, pastoralists indirectly decide on a stocking rate, that is the

herd size, as extraction directly affects the number of herbivores in equilibrium.

Pastoralist income primarily consists of a combination of animal sales, which depend on off-take levels, and products such as milk, blood and wool, which depend on the herd size. Commercially oriented pastoralists rely more on the animal sales, while subsistence-oriented ones rather prefer the income from large herd sizes (Nyariki and Amwata, 2019; Little et al., 2014; Hauck and Rubenstein, 2017; Heron and Gertel, 2011). Additionally, a larger herd size serves as a means to preserve wealth and as an insurance to future shocks (McCabe et al., 2025). Apart from just having an economic purpose, in many regions herd ownership also has a large cultural value. Cattle plays a role in ritual or social ceremonies and can be an indicator of social status. Some ethnic tribes view herd ownership as part of their identity (Adriansen, 2006; Hauck and Rubenstein, 2017). A greater distance to markets or urban areas also affects to which extend pastoralists engage in commercial sales, as it requires effort, time as well as knowledge of markets to effectively make use of them (Little et al., 2014). To which degree pastoralists interact with markets and pastoral livestock strategies depend on prices is therefore determined by a range of factors (Heron and Gertel, 2011).

We model this interaction by having pastoralists receive utility from both selling animals and owning a herd. The importance of animals sales, relative to the importance of herd size, is proxied by means of a relative livestock price parameter, which reflects how a combination of degree of commercialisation, cultural values, ease of market access and actual market prices influence pastoral decisions about off-take.

High stocking rates have been identified to increase the vulnerability of drylands to desertification (Zhao et al., 2005; Hein and Ridder, 2006). This risk is prominent in the face of exogenous shocks such as droughts, which can rapidly deplete the grass cover. The capacity of an ecosystem to remain in, or return to, a desirable state when subjected to such disturbances is commonly referred to as its resilience. Following Holling (1973), we define resilience as the magnitude of disturbance required to shift the system into an alternative regime. We specifically differentiate resilience to shocks which are irreversible — effecting degradation even if the off-take policy is subsequently changed — and reversible by an adequate adjustment of the off-take policy. Still, to recover from a reversible shock, herd size and off-take level might need to be adjusted beyond what is feasible for a stable subsistence level. In practice this may result in a prolonged famine. We refer to resilience to irreversible

shocks as *irreversibility resilience*, and to resilience to reversible shocks as *management resilience*.

We examine how changes in the livestock price proxy, reflecting a combination of market access, market prices and pastoral commercialisation, influence the irreversibility and management resilience of dryland systems under drought conditions, assuming utility-maximizing behavior by pastoralists. In addition, we assess how these resilience measures vary with site-specific abiotic properties such as soil type and rainfall. We differentiate between clayey soils, which are more prone to desertification as water infiltration is low with low grass cover, and sandy soils which allow for some water infiltration even if grass cover is low.

Our results show that irreversibility resilience increases with higher livestock prices, as pastoralists tend to raise off-take rates, resulting in lower stocking densities and greater vegetation cover. This pattern is consistent across a range of rainfall levels and soil types. Management resilience, by contrast, declines with increasing prices and extraction intensity. This indicates that systems exposed to stronger market incentives are more resilient to regime-shifting shocks but less resilient to shocks that require adjustments in extraction strategy. High off-take levels are difficult to sustain and, more importantly, threaten economic livelihoods if pastoralists are unable or unwilling to reduce off-take rapidly when drought occurs.

For systems with more rainfall, management resilience initially increases with rising prices before declining again, exhibiting a non-monotonic response. This relationship holds for both clayey soils with very low water infiltration and sandy soils with higher water infiltration. In clayey soils with substantial rainfall, increased extraction prevents herds from growing too large and depleting the grass cover too much, which prevents sufficient water infiltrating into the ground. As such extraction is actually necessary to maintain a stable equilibrium in which grazing is possible. This implies that under these conditions without anthropogenic off-take, herds cannot be sustained.

Beyond investigating the resilience under the assumption that pastoralists maximise their utility, we mapped the trade-off between utility and resilience by finding the extraction levels that resulted in Pareto optimal combinations of utility and resilience. We found that pastoralists can give up some utility to increase the resilience of the system in a Pareto optimal way. Even more strikingly, we found that under most conditions a small percentage decrease of utility already greatly increases either or both irreversibility and management

resilience. When livestock prices increase, this trade-off becomes sharper and pastoralists have to give up more utility to increase resilience. We also found that resilience and utility do not have to have a trade-off at all: under some conditions they also reinforce each other.

Our analysis aligns with [Vignal et al. \(2023\)](#), who examined how alternative extraction rules affect the resilience of a comparable dryland model. We extend this line of research by explicitly integrating the two main pastoral objectives into the ecological model and by distinguishing between management and irreversibility resilience. Whereas Vignal et al. focused on the ecological consequences of different extraction strategies, our approach incorporates the economic dimension by linking off-take decisions to livestock prices. This allows us to provide deeper insight into how pastoralist behavior—and thus the resilience of dryland systems—is shaped by market conditions.

As noted by [Anderies et al. \(2002\)](#), discounting is not required when the analysis is restricted to long-term equilibrium profits. Constant off-take is representative of a pastoral society that needs to sell or consume some animals each period to maintain a viable subsistence level ([Yatat-Djeumen et al. 2025](#)). It has also been argued that constant, conservative grazing strategies often outperform opportunistic ones, even in variable environments ([Higgins et al. 2007](#)).

Several comparable studies have been conducted on pastoralist decision-making, using a wide range of ecological mechanisms and management strategies. [Stigter and Van Langevelde \(2004\)](#); [Perrings and Walker \(2004\)](#); [Crépin and Lindahl \(2009\)](#) all use a deterministic setting that allows for regime shifts, while [Hein and Weikard \(2008\)](#); [Weikard and Hein \(2011\)](#); [Quaas et al. \(2007\)](#); [Janssen et al. \(2004\)](#) use stochastic dynamics without explicitly modelled regime shifts. Stochastic models capture the year-to-year variance in rainfall observed in these areas, but regime shifts that happen over longer time scales are well described by deterministic models that focus on long-term averages. In [Quaas et al. \(2007\)](#) and [Janssen et al. \(2004\)](#) pastoralists receive profits by production of wool and milk, while in [Stigter and Van Langevelde \(2004\)](#); [Perrings and Walker \(2004\)](#); [Crépin and Lindahl \(2009\)](#); [Weikard and Hein \(2011\)](#) just profits from animal sales are considered. One strand of papers lets the herbivore population develop dynamically with the available vegetation, whereas some studies allow them to be fully controlled by the pastoralists.

A similar separation of resilience and profit was employed by [Anderies et al. \(2002\)](#) in a fire-driven rangeland model; we extend that framework by mapping the full Pareto-efficient frontier of resilience–profit trade-offs.

Economic valuations of resilience began with [Mäler \(2008\)](#), who derived an “accounting price” by linking resilience to a lower probability of regime shifts. Subsequent studies have framed resilience as an insurance value: [Baumgärtner and Strunz \(2014\)](#) quantified the risk reduction provided by resilience based on managers’ risk preferences, and [Derissen et al. \(2011\)](#) showed that resilience can be necessary—though not always sufficient—for sustainable exploitation.

However, these analyses typically assume equilibrium pay-off and resilience are independent. In contrast, we demonstrate that in dryland grazing systems they are interdependent through grazing management choices. The nature of this coupling depends on ecological parameters (e.g., soil water infiltration, rainfall) and economic parameters (e.g., market prices), allowing us to quantify how economic factors affect resilience without explicit stochastic modelling.

The remainder of the paper is organized as follows: Section 2 introduces the ecological-economic model, Section 3 analyzes the system under management, Section 4 explores the resilience–utility trade-off, and Section 5 concludes with a discussion.

2. Model

Ecological dynamics. Our starting point is the ecological model of [van de Koppel and Rietkerk \(2000\)](#), which describes the dynamics of a water-limited dryland system grazed by herbivores. We extend this model by including an extraction term in the herbivore dynamics. The plant and herbivore stock variables are expressed as densities, which describes their average biomass per unit of area. Dynamics are characterised by a positive feedback between water and vegetation, and a predator-prey interaction between herbivores and vegetation. Plant growth is limited by water availability, which in turn is strongly dependent on plant density. The underlying mechanism is the facilitation of water infiltration by the plant roots. The water-plant dynamics equilibrate on a faster time scale than the plant-herbivore dynamics, resulting in water availability being at its steady state value $W^*(P)$ which is a function of the plant density P , of the form

$$W^*(P) = W_{\text{in}} \frac{P + kI_0}{P + k} \frac{1}{uP + r_W}. \quad (1)$$

Here W_{in} is the precipitation, I_0 the minimum water infiltration in the absence of plants, k a half saturation constant, u a coefficient relating specific

uptake of water by plants to water availability and r_W is the rate of water loss due to exogenous effects like evaporation.

Plant (P) and herbivore (H) densities evolve according to

$$\dot{P} = hW^*(P)P - mP - bHP \quad (2a)$$

$$\dot{H} = bgHP - dH - E. \quad (2b)$$

Here m and d are the respective mortality rates of the plants and herbivores, h relates the specific uptake of water by plants to water availability, b is the per capita consumption of plants by herbivores and g is a consumption-to-growth conversion factor: the model assumes a linear relationship for the consumption of plants by herbivores. The mortality rate for vegetation is much higher than the one for herbivores, indicating that vegetation dynamics operate faster than herbivore dynamics.

The values of these ecological parameters as they are used in this paper are detailed in table [1](#). Values are slightly modified from [Rietkerk et al. \(1997a\)](#) and [van de Koppel and Rietkerk \(2000\)](#), who base them on earlier experimental work done by [van Wijngaarden \(1985\)](#). Most parameters were scaled in order to have plant and herbivore stocks in equilibrium similar to those found in literature ([Gebremedhn et al., 2023](#); [Dixon et al., 2020](#); [Breman and de Wit, 1983](#)).

We use the linear functional response bPH to model herbivore grazing. Consumption by livestock is measured to be increasing almost linearly for low plant densities while saturating at higher plant densities [Short \(1985\)](#). In our areas of interest vegetation density stays within the linear regime, so it is not necessary to use a saturating linear response function. The extraction term $E \geq 0$ represents the off-take of animals for slaughter, self-consumption or trade. Many rangeland models allow for negative extraction, that is, the purchase of extra animals, but we assume this is not possible for subsistence pastoralists with low savings and difficult access to markets.

Economic decisions. In the present paper, we model the pastoralists as a single cooperative decision maker that aims at maximising a common objective. This is not an unusual arrangement ([Gulelat, 2002](#)). We defer the analysis of other decision structures to future work.

Our system is fully deterministic, and pastoralists are only able to set constant extraction rates. This simplification allows us to look at the long-term behavior of the system. In dryland systems, rainfall can be highly

Parameter	Description	Value	Unit	Dimension
h	coefficient relating specific plant growth to water availability	0.0015	$[m^2mm^{-1}]$	$\frac{1}{[W][T]}$
m	specific loss of plant biomass	0.005	$[d^{-1}]$	$\frac{1}{[T]}$
b	per capita consumption by herbivores	0.0004	$[m^2d^{-1}g^{-1}]$	$\frac{1}{[H][T]}$
g	consumption-to-growth conversion coefficient	0.2	$[g^{-1}]$	$\frac{[H]}{[P]}$
d	mortality rate of herbivores	0.002	$[d^{-1}]$	$\frac{1}{[T]}$
W_{in}	rainfall	0 – 3	$[mm d^{-1}]$	$\frac{[W]}{[T]}$
I_0	minimum water infiltration in the absence of plants, expressed as proportion of rainfall	0 – 1	—	\square
u	coefficient relating specific uptake of water by plants to water availability	0.002	—	$\frac{1}{[P][T]}$
k	half saturation constant	25	$[g m^2]$	$[P]$
r_w	rate of water loss	0.1	$[d^{-1}]$	$\frac{1}{[T]}$

Table 1: Ecological parameters

variable from year to year, but averaged over decades it is relatively constant. Given that we consider the dynamics over a long time period, as long as disturbances are small enough to not cause a transition to another steady state, the system has enough time to return to the original steady state. Accordingly, we examine pastoralist benefits and utility in the steady-state setting. We do, however, discuss the resilience of these steady states below. We also note that in steady state, the dynamically optimal extraction rate will be constant.

Pastoralists derive utility from two main sources: the sale of harvested animals and ownership of a herd. The latter is generated by sale and consumption of animal products such as milk, blood or wool, and the fact that in many societies livestock ownership serves as a form of insurance and a symbol of social status (Adriansen, 2006). Assuming utility to be linear in both animal sales and herd ownership (Quaas et al., 2007; Stigter and Van Langevelde, 2004) yields

$$U = p_c E + qH, \tag{3}$$

where U is the steady-state benefit stream. We omit discounting and more complex utility structures as our analysis focuses on equilibrium outcomes (Anderies et al., 2002). We also abstract from costs associated with either harvesting or maintaining herds; pastoralists are constrained only by the ecological capacity of the system. We normalize units by setting q , the marginal utility gained by owning an extra unit of animals, equal to 1. The parameter p_c can be interpreted as the pastoralists' valuation of monetary revenue from selling animals on the livestock market relatively to keeping animals for their ongoing products or social functions. Accordingly, p_c serves as an indicator of the degree of commercial orientation. Pastoralist groups that primarily engage in commercial livestock production are characterised by high values of p_c , while those who rely on herding for household subsistence are described by lower values of p_c . More commercial groups will produce more if prices are higher, and some groups who are less commercially oriented might still be convinced to sell some animals if the prices are high (Heron and Gertel, 2011). An alternative interpretation of p_c is ease of access to livestock markets. Pastoralists at large distances from markets will incur travel costs; moreover, they are under more pressure making sales quickly and they have fewer opportunities to acquire price information before travelling (Little et al., 2014).

2.1. Nullclines and Steady States

The present section analyses the plant–herbivore system (2) under time-constant harvesting. For a full classification of the dynamics of the grazing system without harvesting of the herd we refer to van de Koppel and Rietkerk (2000) and to Langelaan et al. (2026). Without harvesting the model endogenously generates cyclical behaviour between periods with an abundance of plants and herbivores, and periods where the biomass of both is much reduced. We do not observe this behavior under constant harvesting, but we do still observe bi-stability between multiple stable steady states, defining qualitatively different ecological regimes.

First we give some general information about the nullclines and steady states of the system before we discuss the full classification.

Steady states. We introduce the plant growth rate

$$r_P(P) = hW^*(P) - m.$$

The function $r_P(P)$ takes its maximum at a single point $P_{\max} > 0$. We only consider the parameter sets that allow for positive plant growth, i.e. $r_P(P_{\max}) > 0$. The plant growth rate in the absence of any plants, $r_P(0) = \frac{hW_{in}I_0}{r_w} - m$, determines whether the system can undergo irreversible or reversible changes. For the purpose of this paper, we call systems for which $r_P(0) < 0$ *irreversible*, as they have the possibility to irreversibly degrade into a desert state where plants are unable to spread over barren soil. If $r_P(0) > 0$ plants can proliferate over barren soil and we call these systems *reversible*. See figure 1.

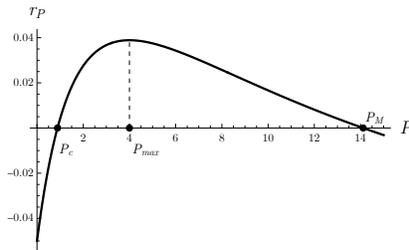


Figure 1: Plant growth rate, arid grazing system

For irreversible systems, characterized by $r_P(0) < 0 < r_P(P_{\max})$, the growth rate has a single zero P_c in the interval between 0 and P_{\max} . The

growth rate also has a single zero P_M in the interval that runs from P_{\max} to infinity: this is the carrying capacity of the grazing system, that is, the steady state grass biomass if there are no herbivores.

Introducing the plant growth rate allows us to rewrite the plant dynamics (2a) as:

$$\dot{P} = r_P(P)P - bHP = (r_P(P) - bH)P \quad (4)$$

Plant biomass growth is zero, that is $\dot{P} = 0$, if (P, H) is on the first nullcline

$$N_1 : P = 0 \quad \text{or} \quad H = r_P(P)/b. \quad (5)$$

Similarly, the herbivore dynamics, given by equation (2b), implies that for positive extraction, $E > 0$, herbivore biomass growth is zero on the second nullcline

$$N_2 : H = \frac{E}{bgP - d}. \quad (6)$$

Under positive extraction, there are therefore either two or no steady states with positive plant and herbivore biomasses, depending on whether nullclines N_1 and N_2 intersect or not: we denote these intersections as (\bar{P}_E, \bar{H}_E) and (P_m, H_m) , see figure 2. We find that the maximal grass biomass state under extraction state (P_m, H_m) is always a saddle, while the grazing state (\bar{P}_E, \bar{H}_E) is either stable or unstable, depending on conditions and choice of extraction.

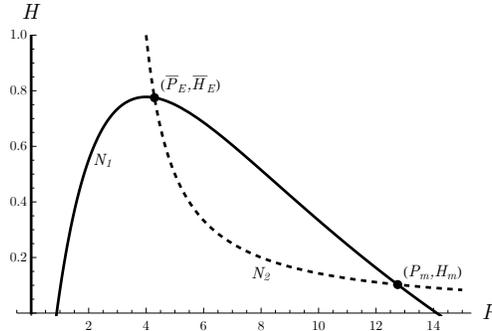


Figure 2: Nullclines under Constant Extraction

A constant positive extraction rate $E > 0$ cannot be sustained at the boundary $H = 0$, as positive extraction is not possible from a depleted herd. Therefore, we have to require $E = 0$ if $H = 0$: at the boundary, the dynamics

reduces to pure plant dynamics. This means that for irreversible systems, the degraded state $(P, H) = (0, 0)$ is a stable steady state, as whenever the plant population drops below P_c , deterioration of the plant stock cannot be prevented by fully removing the herds. Alleviating grazing pressure by increasing the extraction does not prevent tipping in this case, as the plant density is already too low to enable sufficient water infiltration for new plant growth.

Mandating $E = 0$ at the boundary also means that if $P_c > 0$ and $P_M > 0$, the boundary points $(P_c, 0)$ and $(P_M, 0)$ are steady states, which are respectively locally unstable and locally stable with respect to the plant-only dynamics. We repeat that only irreversible systems have positive critical points $P_c > 0$. Very dry systems have $P_M < 0$, which means only the desert steady state exists.

3. Stability and Resilience

3.1. Classification

We outline the various equilibrium configurations of the system that arise from differences in site-specific characteristics (e.g., soil type and rainfall) and from alternative choices of the extraction level E . As seen in figure 2, internal equilibria only exist when nullclines N_1 and N_2 intersect. As E increases, the nullcline N_2 shifts upwards, until it no longer intersects N_1 : that is, if the level of extraction E is too high, there is no internal equilibrium. The maximum extraction that still allows for a steady state is the one that causes these nullclines to have exactly one common point, i.e., the nullclines are tangent. This extraction level is the maximum sustainable yield (MSY), which has earlier been introduced for fishery models but has also been applied to grazing systems (Barlow, 1987). For extraction levels below the maximum sustainable yield there exist two internal steady states: the grazing state (P_E, H_E) which is either an attractor or a repeller and the maximum vegetation state (P_m, H_m) which is always a saddle. With regards to the main issue, namely the existence and stability of the grazing steady state, we therefore identify three main configurations.

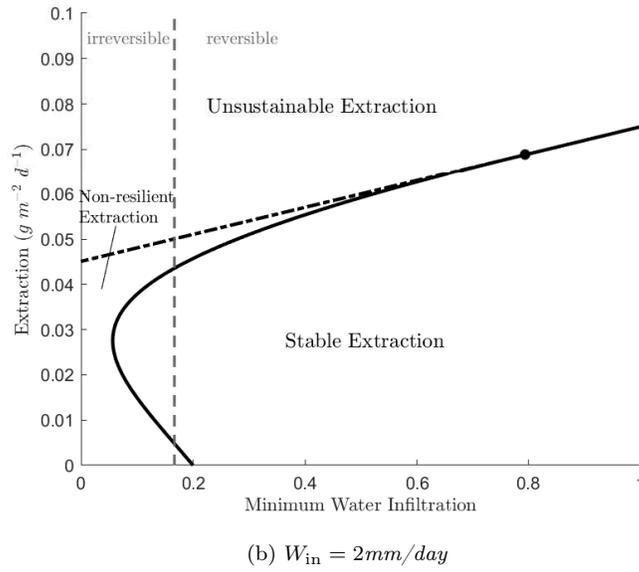
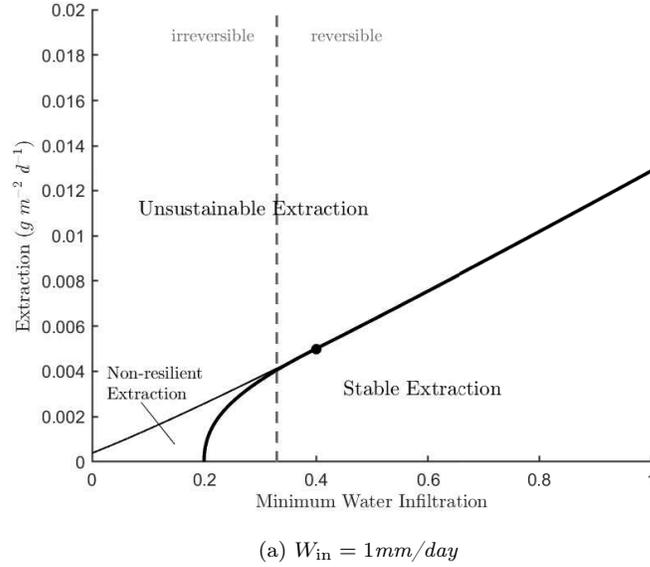


Figure 3: Minimum water infiltration vs extraction for two levels of rainfall W_{in} . Left of the dashed grey line the system is irreversible. For the situation with higher rainfall ($W_{\text{in}} = 2 \text{ mm/day}$), there are soil conditions for which a non-zero extraction is necessary to stabilize the system, as the stable extraction region has a hump.

- **Unsustainable Extraction.** There are no viable internal equilibria:

all trajectories converge to zero herd size, i.e. the boundary $H = 0$. This happens when extraction is higher than the maximum viable yield.

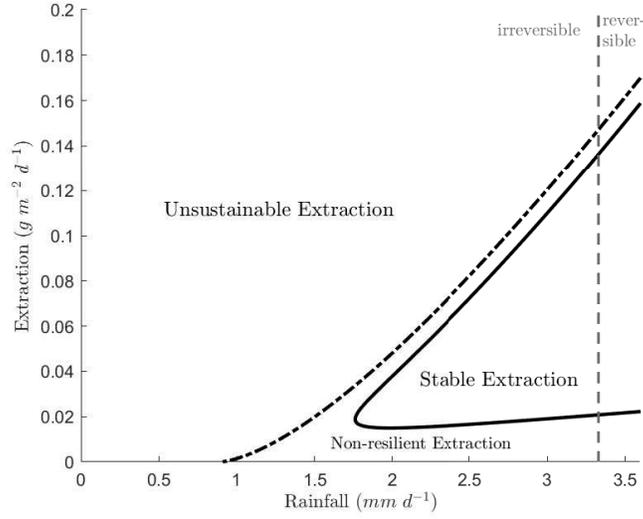
- **Non-resilient Extraction.** The grazing state is locally unstable, so theoretically long-term extraction is possible, but it can easily be disrupted in which case the herd size converges to zero.
- **Stable Extraction.** The grazing steady state is locally stable. It is not the sole attractor of the system: trajectories outside of its basin of attraction converge to a zero herd size.

Figure 3 shows which levels of extraction and base water infiltration I_0 result in one of the three aforementioned configurations for a given level of precipitation, in this case $W_{\text{in}} = 5$. Also indicated is for which base infiltration levels the system is reversible or irreversible. Phase diagrams with sample trajectories starting from various initial conditions illustrating possible evolutions of the plant and herbivore stocks are shown for each configuration in figure 5. The basin of attraction of the locally stable grazing state is indicated by a dashed line. If the initial plant and herbivore stocks are such that they are located within this basin of attraction, the chosen extraction level will result in a trajectory towards the steady state. Outside of this basin of attraction, all trajectories converge to the boundary $H = 0$.

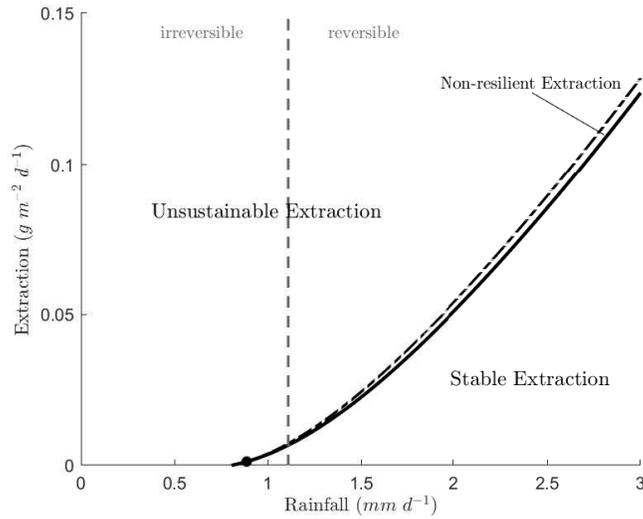
When herds are completely depleted, extraction is no longer possible and only the plant dynamics without grazing are relevant. For reversible systems, this means that the plant density will recover to the maximal vegetation steady state P_M after the herds are gone, as plants can regrow even at low densities. If the system is irreversible, whether the system fully degrades depends on the state of the plant cover after the depletion of the herds. If plants are less abundant than the critical density P_c , the system will degrade to the desert equilibrium without any plants or herbivores, while if enough plants are left, i.e. $P > P_c$, the plant population will recover to P_M in a similar way to reversible systems. The difference matters: if the system ends up with a non-zero plant biomass, a herbivore population can always be reintroduced at a later time by acquiring animals from outside the system and economic viability could theoretically be restored. However if the system is irreversibly degraded, no such restoration effort can be made.

The saddle equilibrium only attracts trajectories whose initial conditions of plant and herbivore densities are located exactly on the stable saddle-paths. Small variations on this path cause the trajectory to converge to a

different state, so in practice it is unlikely the system will end up in the saddle state.



(a) $I_0 = 0.1$, Clayey Soil.



(b) $I_0 = 0.3$, Sandy Soil.

Figure 4: Rainfall (mm/day) versus extraction ($grams\ of\ biomass\ m^{-2}/day$) for two levels of minimum water infiltration (I_0). Clayey soils allows less water infiltration ($I_0 = 0.1$), while sandy soils allow more water infiltration ($I_0 = 0.3$). For sandy soils, left of the dashed grey line, the system is irreversible. Clayey soils are always vulnerable for these rainfall levels, as the threshold is located at a rainfall level of $3.33\ mm/day$.

For dryland grazing systems, sandy soils allow for a higher water infiltration, while clayey soils allow for less water infiltration [Rietkerk et al. \(1997b\)](#). Figure 4 shows the classification diagram when the soil type I_0 is constant and both extraction and rainfall are varied. We obtain the striking result that for clayey soils, which are generally irreversible, a positive extraction is necessary to even exhibit a stable grazing equilibrium. Without human management, the herbivore biomass is not stably viable, and high initial stocks might even degrade the system irreversibly.

More sandy soils are reversible for a larger range of rainfall levels as can be seen in the second panel of figure 4. If a grazing equilibrium is possible under extraction, it is almost automatically also stable, as the non-resilient extraction configuration only includes a small range of parameters.

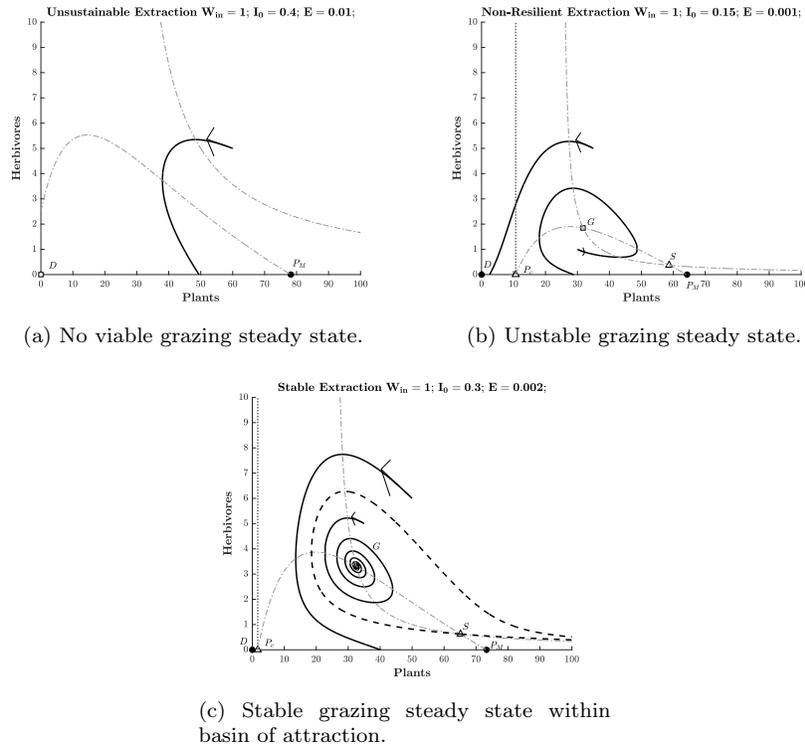


Figure 5: Phase Diagrams of the three configurations. Sample trajectories are indicated with solid lines, nullclines with grey dash-dot lines and the vulnerability frontier with a grey dotted line. For (c), the boundary of basin of attraction of the stable grazing state is indicated with a dashed black line.

3.2. Resilience

Now that we have mapped under which conditions grazing under extraction is stable, we aim to give some measure of the resilience of this stable grazing state. We use the definition of ecological resilience as introduced by [Holling \(1973\)](#). This refers to the extent of a shock that a system can absorb while still returning to its original steady state. For one-dimensional systems this is the distance between the steady state and the boundary of its basin of attraction. For systems with higher-dimensional state spaces, the geometrical minimum distance to the boundary of the basin of attraction in any direction has been used as a resilience measure [Anderies et al. \(2002\)](#); [Krakovská et al. \(2024\)](#); [Dakos and Kéfi \(2022\)](#).

In our context, we consider shocks that affect only one population — either plant density or herbivore density — since we expect that ecological shocks (such as diseases affecting cattle or droughts affecting vegetation) typically impact only one of these populations at a time. Droughts generally cause a collapse of both the the plant and herbivore stock, but the excess mortality of herbivores is mostly because of malnutrition, which is already endogenously captured in our model [Catley et al. \(2014\)](#). We call these H-shocks and P-shocks respectively. Additionally, we focus on shocks that affect the system in the negative direction, that is, shocks that reduce stock levels. Our resilience measure is hence the distance between the steady state stock level and the boundary of its basin of attraction in a single dimension, in the negative direction. Additionally we make a distinction to what we call *Irreversibility Resilience* and *Management Resilience*, which is explained below.

P-shocks that bring the plant stock from the steady state to below the critical plant cover P_c will irreversibly degrade the system. We hence define the difference of the steady state plant cover to the critical cover $R_I \equiv P_E - P_c$ as the *Irreversibility Resilience*. The system is resilient to irreversible change as long as shocks are smaller or equal to its irreversibility resilience. See figure [6](#).

If a P-shock shifts the state variables beyond the basin of attraction of the steady state (P_E, H_E) , yet the plant cover does not fall below P_c , an appropriate policy change can return the system to the steady state. That is, outside of the basin of attraction of the steady state, pastoralists could apply a different extraction rate — possibly negative or time-varying — to move the system back to the steady state. We hence call the distance in the negative P-direction from the steady state to the boundary of its basin of

attraction the management resilience of the chosen policy. The management strategy, that is the chosen extraction level, is resilient to shocks up to this value. For a shock larger than the management resilience, the policy has to be changed for the system to sustainably allow economic activity, i.e. to have a stable non-zero herbivore population.¹

H-shocks only affect the herbivore stock and do not affect the plant density directly. Consequently, H-shocks never cause the plant density to fall below P_c and hence never irreversibly degrade the system. H-shocks do cause a depletion of the herd if pastoralists do not change their extraction rate, but the plant density will converge to the maximal vegetation state P_M . As there are no shocks to the herbivore population that can cause irreversible changes, there exists no irreversibility resilience in the H-direction and we just consider management resilience in the H-direction, as seen in figure 6.

The kinds of resilience we therefore consider are: irreversibility resilience in the P-direction, management resilience in the P-direction and lastly management resilience in the H-direction. While our study focuses on resilience with respect to shocks, we do not explicitly model the probability of different shock sizes or their arrival rate. This is in line with the focus, in this paper, on the asymptotic, rather than the transient, system behaviour. Figure 6 visualizes the shocks and corresponding resiliencies that we consider in our analysis. Future work could explore how shock probabilities interact with resilience measures to provide a more comprehensive understanding of system vulnerability.

4. Pareto Optimal Resilience and Profits

4.1. Utility-Resilience trade-off

Figure 7 illustrates how management resilience and irreversibility resilience respond to changes in extraction level. Irreversibility resilience increases with extraction, as higher extraction levels lead to equilibria with fewer herbivores and consequently greater plant biomass, without altering the

¹To be precise: if a shock brings the system in a state from which it then is on a trajectory that would cause it to drop below P_c , pastoralists would increase the extraction to alleviate the grazing pressure to boost plant growth, while later reducing the extraction again to allow the herd to recover. If the system would be shocked to a state in which just the herd would converge to zero, decreasing the extraction immediately should allow the herd enough respite to regrow.

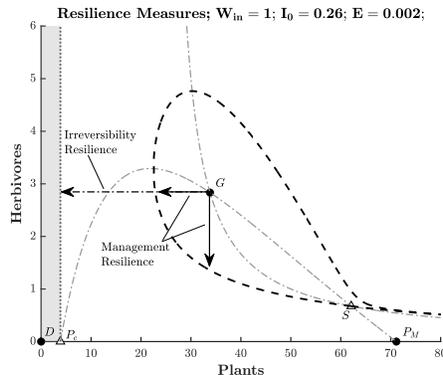


Figure 6: Irreversibility Resilience and Management Resilience in an irreversible system. The dashed line marks the boundary of the basin of attraction of the steady state G . The grey region indicates the initial conditions that lead to desertification under any extraction policy.

critical plant biomass at which irreversible systems collapse to a desert state. The effect of extraction on management resilience, however, is site-specific. We consider three representative cases: two irreversible systems—one with low rainfall and medium water infiltration (a) and another with high rainfall and low water infiltration (b)—and one reversible system characterized by high rainfall and medium water infiltration (c).

For irreversible systems with low water infiltration (panel b, Figure 7), management resilience exhibits a non-monotonic response to extraction, with an internal maximum. In systems with higher infiltration and lower rainfall (panel a), management resilience declines monotonically with extraction. For reversible systems (panel c), management resilience again displays a non-monotonic profile with an internal maximum. These patterns indicate that, if managers are aware of site-specific characteristics, extraction levels can be adjusted to maximize system resilience. In all cases, increasing extraction to the highest level that still supports a stable steady state results in minimal management resilience.

Whether such an extraction strategy is economically optimal depends on the aggregate benefit streams at that extraction level. Utility from livestock sales increases linearly with extraction, whereas herd utility—which directly reflects herd size—declines with extraction. Hence, herd utility is maximized under minimal extraction, while extraction utility peaks at maximal extraction. The extraction level that maximizes total utility therefore depends on

the normalized livestock price p_c , which governs the slope of the extraction utility function.

As noted by Perrings and Walker (2004), moderate levels of exploitation can enhance the resilience of grazing systems. Comparable visualisations of resilience–utility relationships have been used in other models, such as that of Anderies et al. (2002).

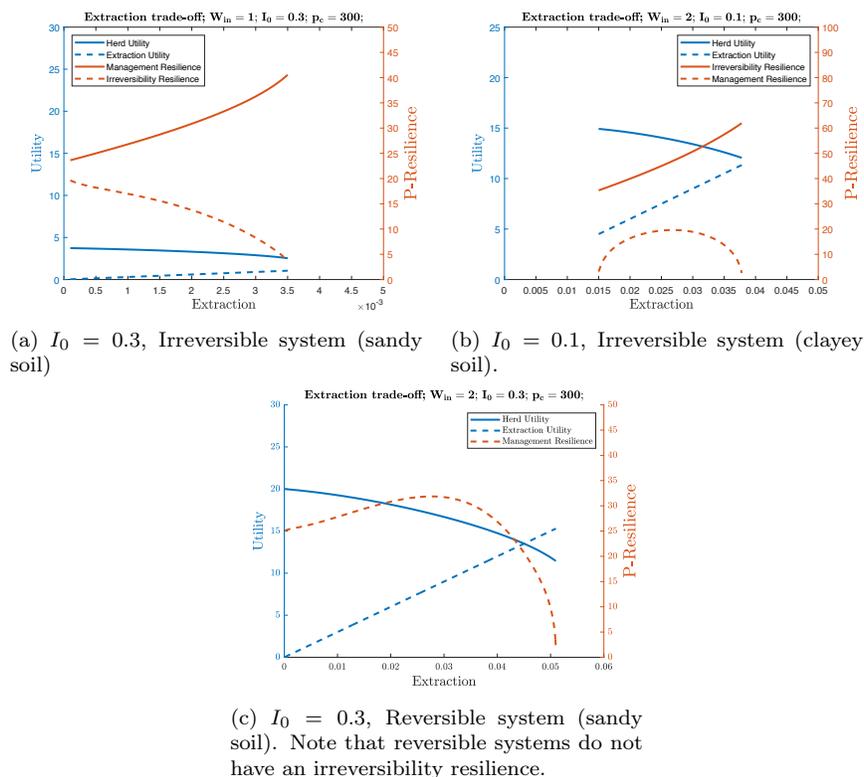


Figure 7: P-resilience and utility as function of extraction.

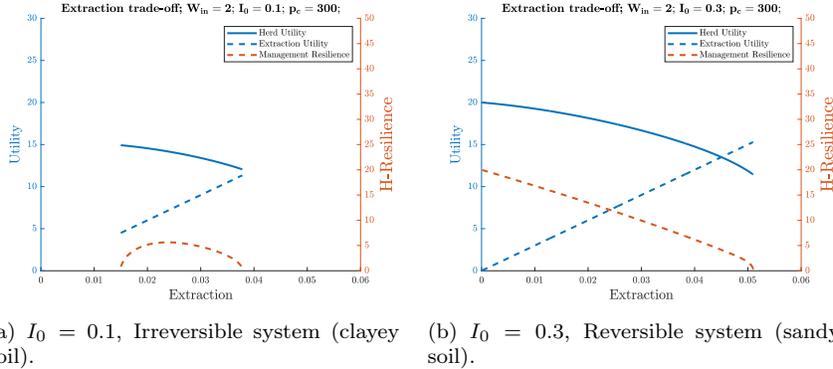


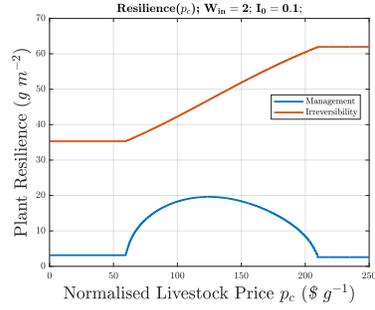
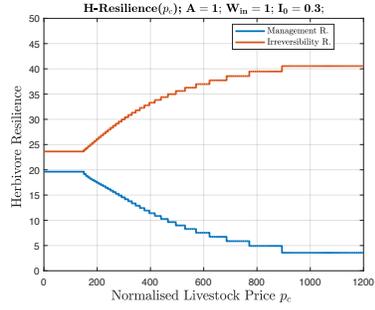
Figure 8: H-resilience and utility as function of extraction.

4.2. Resilience as function of price for utility maximizing pastoralists

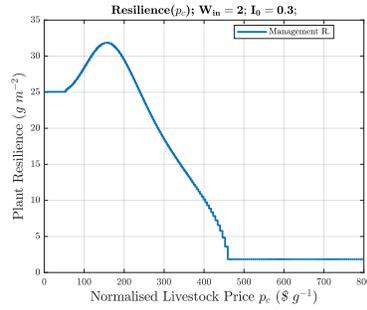
Assuming that pastoralists choose the extraction level that maximizes utility for a given livestock price, it is possible to infer a relationship between resilience and price. Figure 9 illustrates this relationship for P-resilience. Higher livestock prices increase irreversibility P-resilience, making the system more resistant to collapse-inducing shocks. In contrast, management P-resilience tends to decline with increasing prices. However, in cases with higher rainfall—illustrated in panels (b) and (c)—an increase in price can initially enhance management resilience. This positive effect persists only up to a threshold price level, beyond which resilience declines again.

Higher prices also widen the gap between irreversibility and management resilience. Consequently, the ecological system itself becomes more resilient, while the robustness of the extraction policy decreases. While irreversible changes to the ecosystem are unwanted, reversible changes can also have severe consequences. When herds disappear, due to a shock greater than the management resilience, it can lead to famine, even if herbivores could successfully be reintroduced at a later point in time. Therefore, for local populations, management resilience can be as, if not more, important as irreversibility resilience.

Figure 10 shows that moderate price increases, which incentivize higher extraction, can enhance management H-resilience in irreversible systems. In contrast, such price increases do not yield similar benefits for reversible systems.

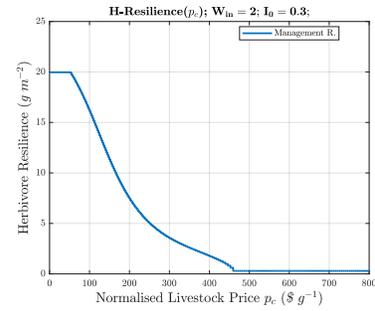
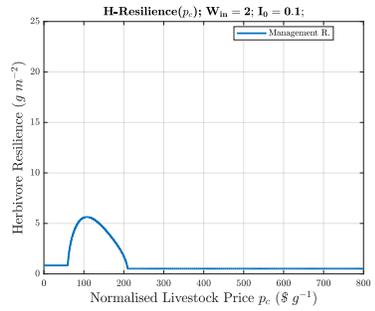


(a) $I_0 = 0.3$, Irreversible system (sandy soil) (b) $I_0 = 0.1$, Irreversible system (clayey soil).



(c) $I_0 = 0.3$, Reversible system (sandy soil)

Figure 9



(a) $I_0 = 0.1$, Irreversible system (clayey soil). (b) $I_0 = 0.3$, Reversible system (sandy soil).

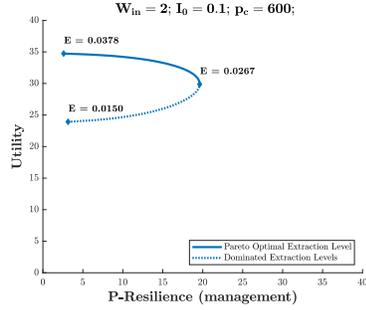
Figure 10

4.3. Utility-Resilience Pareto Frontiers

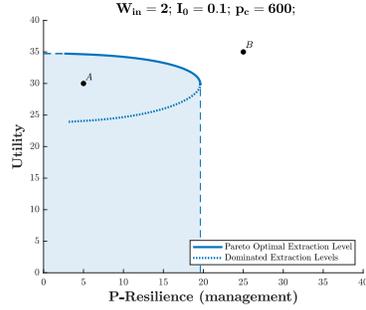
In the previous subsection, we examined the relationship between resilience and livestock prices for utility-maximizing pastoralists. As shown in Figure 7, a trade-off exists between different forms of resilience and utility. Figure 11 plots utility versus resilience for all viable extraction levels. It becomes clear that some extraction levels result in combinations of utility and resilience which are Pareto optimal, that is the values of utility and resilience cannot both be increased at the same time by varying the extraction. The extraction levels that yield Pareto-efficient combinations of resilience and utility form a Pareto frontier. It is possible that only a single extraction level is Pareto optimal, in that case the Pareto frontier consists of a single point, see for example panel (c) of figure 11. Any combinations with a lower resilience or utility than those on the Pareto frontier are minimally achievable, see panels (b) and (d) of figure 11, while any outside are not achievable at all.

Figures 12 and 13 map Pareto frontiers for different site-specific properties rainfall W_{in} , soil type I_0 and extraction utility factor p_c . Dashed lines indicate the maximum values of utility and resilience that can be achieved. For P-resilience, both management and irreversibility resilience are shown.

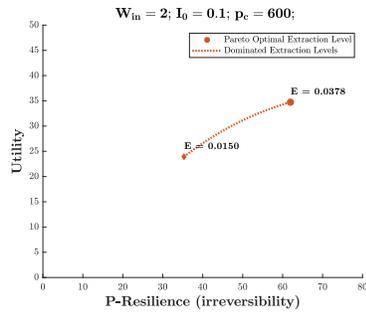
Figure 12 shows that at high prices, the Pareto front for management P-resilience has a negative slope. This implies that managers can increase management resilience only at the expense of utility—or equivalently, increase utility by reducing resilience. Regardless of site-specific properties, the slope of the Pareto front becomes steeper at higher prices, indicating that managers must forgo relatively more utility to achieve marginal gains in resilience.



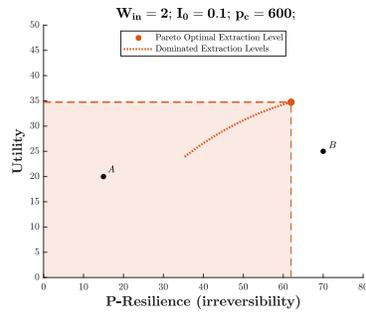
(a) Feasible management resilience/utility pairs



(b) Minimally achievable values (management resilience)

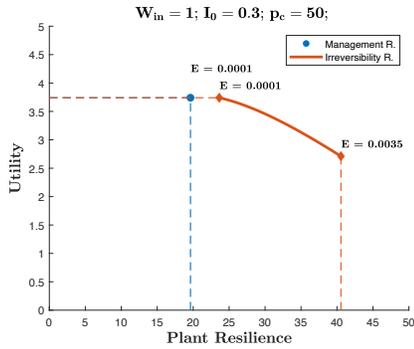


(c) Feasible irreversibility resilience/utility pairs

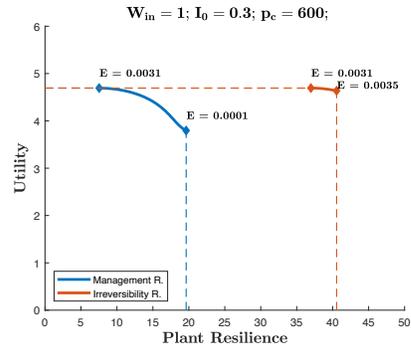


(d) Minimum achievable values (irreversibility resilience)

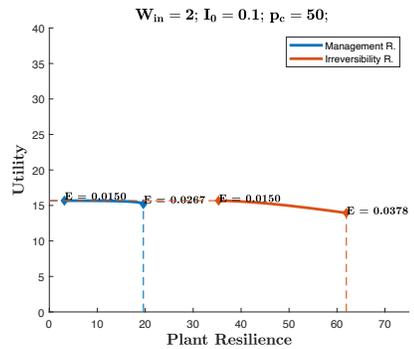
Figure 11: All feasible combinations of utility and management/irreversibility resilience for livestock price $p_c = 600$ are shown in panels (a) and (c) respectively. Pareto optimal combinations are indicated by a solid line, while non-optimal combinations are indicated by a dotted line. Values of utility and resilience within the area enclosed by the Pareto frontiers in panels (b) and (d), such as point *A*, can be achieved minimally. Points outside this region, such as point *B*, can never be achieved. For the irreversibility resilience in panels (c) and (d), only a single extraction level is Pareto optimal, which leads to the minimally achievable region being a rectangle.



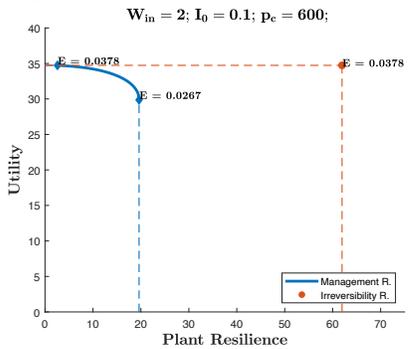
(a) Irreversible system: low rainfall, sandy soil, low price.



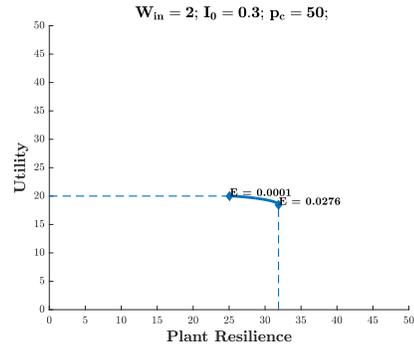
(b) Irreversible system: low rainfall, sandy soil, high price.



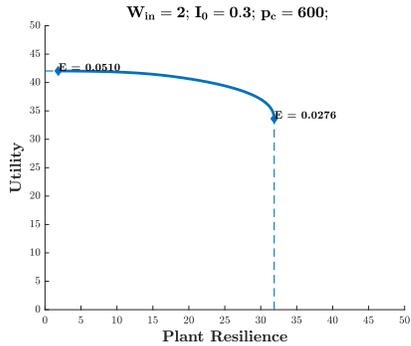
(c) Irreversible system: high rainfall, clayey soil, low price.



(d) Irreversible system: high rainfall, clayey soil, high price.



(e) Reversible system: high rainfall, sandy soil, low price.



(f) Reversible system: high rainfall, sandy soil, high price.

Figure 12: Utility- P-Resilience Pareto fronts for different soil types and livestock prices. Medium rainfall ($W_{in} = 5$). Note that scales differ between clayey and sandy soils, as the latter support steady states with much larger stocks.

Considering irreversibility resilience, at high prices only the maximum

extraction level is Pareto-optimal. This is favourable for managers focused solely on irreversibility resilience, as a single policy simultaneously maximizes both utility and resilience. At low prices, however, the irreversibility-resilience frontier slopes downward: herd utility becomes relatively more important, so increasing utility by reducing extraction leads to lower irreversibility resilience.

Only in panel (c)—the case of an irreversible system with high rainfall, clayey soil, and low prices—is it possible to select an extraction level that is not utility-maximizing but is Pareto-optimal for both management and irreversibility resilience. Under all other conditions (panels a, b, and d), only one form of resilience can be Pareto-efficiently improved at the cost of utility, which simultaneously results in a non-Pareto-optimal level of the other form of resilience at that utility level.

Figure 13 presents the Pareto frontiers for management resilience in the herbivore direction and utility. Similar to management resilience in the plant direction, the slope of the Pareto front becomes steeper at higher prices, and this effect is more pronounced in reversible systems. Although favourable environmental conditions allow for larger herds and greater extraction, this also reduces the basin of attraction, making the system more susceptible to disturbances that can drive it out of equilibrium.

Figure 14 shows the same Pareto fronts as in figures 12 and 13 normalised to the maximum achievable values of utility and management resilience. It is clear that the trade-off between H-resilience and utility becomes much steeper for higher prices, but also for less vulnerable systems. The trade-off between P-resilience and utility is relatively flat, also for higher prices. For P-resilience we see a different effect with respect to variation in site-specific properties, where systems with low rainfall and medium water infiltration (green line in panel (b), figure 14) have a steeper trade-off than less vulnerable systems with higher rainfall (compare yellow line). This indicates that both site-specific properties and market forces are important for the realisation of resilient grazing strategies.

Perhaps the most striking conclusion that can be drawn from these Pareto frontiers, is that they are relatively flat. This means that pastoralists only have to give up little utility to gain a relatively large increase in resilience. This mechanism could explain why some pastoralist societies do already boast considerable resilience (Heron and Gertel, 2011).

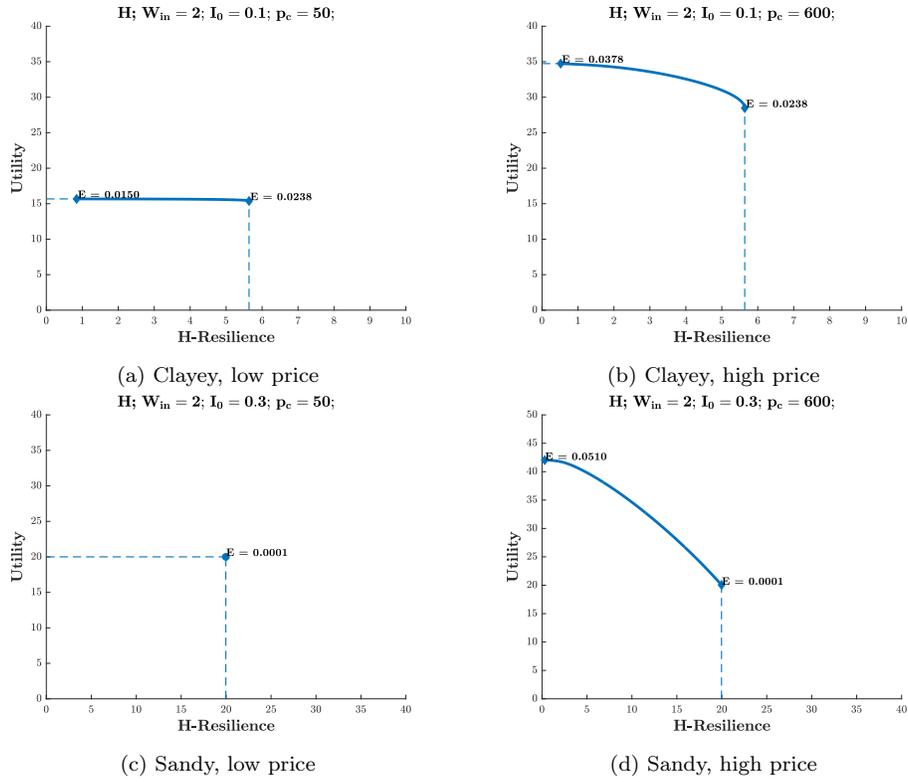


Figure 13: Utility- H-Resilience Pareto fronts for different soil types and livestock prices. Medium rainfall ($W_{in} = 5$). Note that scales differ between clayey and sandy soils, as the latter support steady states with much larger stocks.

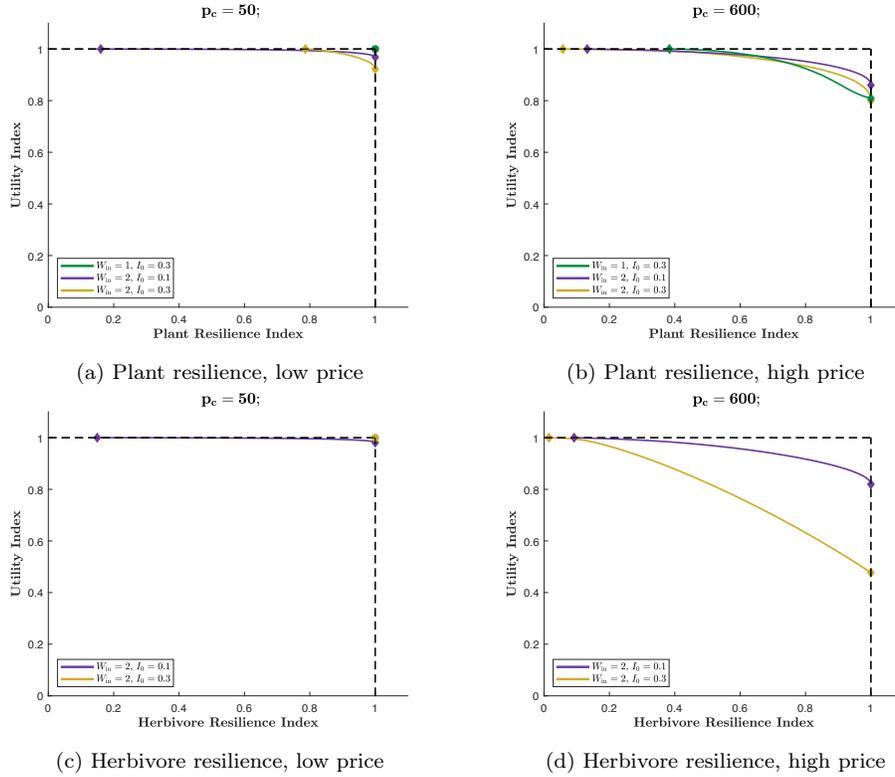


Figure 14: Normalised Utility- Resilience Pareto fronts for two different price levels. Maximal achievable utility and resilience for each different site-specific parameter combination is set to 1. Pareto fronts indicate the fraction of the maximal stock of utility or resilience one has to give up to increase the other stock. Management resilience in the P-direction in panels (a) and (b), management resilience in the H-direction in panels (c) and (d).

5. Discussion

This paper presents two main results. Firstly, we show that changes in livestock prices influence both irreversibility resilience and management resilience through pastoral extraction decisions. Irreversibility resilience increases at higher livestock prices, while management resilience rises with moderate price increases but declines at higher price levels. Secondly, we identify the extraction levels that lead to Pareto-optimal combinations of utility and resilience. Under most conditions, utility and resilience can be traded off, and the steepness of this trade-off depends on livestock prices. Not all pastoralist systems are equally responsive to market prices, so our prices can also be interpreted as the geographical and cultural distance to

markets. Higher prices model pastoralist groups which are more inclined to commercial production of livestock and have better access to markets, while lower prices model those groups which focus on maximising their herd size for utility, rather than trade of animals.

Our analysis goes beyond the work of [Vignal et al. \(2023\)](#), who examined how alternative extraction rules affect resilience in a comparable dryland vegetation–herbivore model. Here, we incorporate the utility streams of pastoral households and distinguish explicitly between management and irreversibility resilience. This adds an economic dimension to the discussion by linking off-take decisions to livestock prices, thereby revealing how market conditions and preferences shape the resilience of coupled social–ecological systems. In regions such as coastal West Africa, where demand for meat is rising, high prices may stimulate greater off-take ([Turner and Williams, 2002](#); [OECD, 2025](#)). In contrast, in Mongolia, where income relies more on products like cashmere and where cultural value is attached to herd size, economic incentives stimulate higher herd sizes ([Berger et al., 2013](#); [Mearns, 2004](#)).

Pastoral systems have long been described as highly adaptive, using herd mobility and resource diversification to buffer ecological variability ([Adriansen, 2005](#); [Freier et al., 2014](#)). In our model, extraction primarily represents the removal of animals from the herd by market sales or consumption. Increasing extraction improves resilience to regime-shifting shocks by reducing grazing pressure. For further research, modifications to the dynamic equations and the utility function could be made to allow the incorporation of spatial redistribution of grazing pressure through herd movement. The way herbivore spatial redistribution is organised is heterogenous across different pastoral systems, ranging from full nomadism to transhumance. Future work could explore how the resilience of one such system depends on further market incentives, preferences and heterogeneity between pastoralists ([Turner et al., 2014](#); [Hauck and Rubenstein, 2017](#)).

Our findings on herbivore shocks align with observations from field studies. Pastoralists with larger herds are often more resilient because they can sell part of their livestock during droughts to purchase supplementary feed while maintaining a viable herd size ([Moritz et al., 2017](#); [Ouédraogo et al., 2021](#)). In our model, larger herds similarly enhance resilience to herbivore shocks, providing a buffer against mortality losses during dry periods.

The model has several limitations. Constant harvesting represents a rigid management strategy. Allowing adaptive harvesting, where off-take responds dynamically to vegetation or herd size, could better capture subsistence main-

tenance under stress. The current framework also assumes equilibrium conditions and does not include inter-temporal trade-offs. Incorporating discounting and dynamic optimisation could improve understanding of how short-term economic pressures interact with long-term ecological stability. The concept of management resilience would benefit from the introduction of discounting, as it enables evaluation of the discounted value loss when a management strategy has to be adjusted on a trajectory back to an equilibrium.

A long-standing debate surrounds whether equilibrium models are suitable for representing dryland systems, which are characterised by high variability and extreme events. Empirical and modelling studies from the Sahel have suggested that excessive stocking rates have accelerated land degradation and overgrazing (Hein and Ridder, 2006; Rahimi et al., 2021), while others have challenged this view (Descroix et al., 2024). For a full overview of this discussion we refer to Vetter (2005); McCabe et al. (2025). Our model follows the equilibrium tradition as it allows for a tractable identification of the relationships between resilience and economic and cultural preferences. Additionally, it has been argued that not all drylands have sufficient high variation in rainfall for non-equilibrium theory to be applicable (Turner and Schlecht, 2019). Our model, being an equilibrium model, shows that extraction decisions mediated by markets and cultural preferences influence both the thresholds of regime shifts and the capacity for recovery. Higher extraction may lower stocking rates and thus reduce vegetation pressure, but it also erodes the ability to adjust when shocks occur, conclusions that also align with non-equilibrium theory.

Our mean-field approach omits spatial and temporal variability such as seasonality, inter-annual rainfall fluctuations, and vegetation self-organisation. This last factor is known to enhance resilience by enabling local recovery and pattern formation (Rietkerk et al., 2021). However, we hypothesize that including it would not alter our results, as grazing by herbivory dampens the extra resilience by vegetation self-organisation (van de Koppel et al., 2002). Our way of incorporating shocks already models major variation in rainfall, and we think adding seasonality or smaller year-by-year variation would not alter our results significantly, as they would not affect the key mechanisms governing the relationships between resilience and pastoralist preferences.

Finally, we treat pastoral utility and ecosystem resilience as separate outcomes. Other studies have included resilience directly in the utility function by weighting it as part of welfare (Baumgärtner and Strunz, 2014; Derissen

et al., 2011). By keeping utility and resilience distinct, we are able to show how preferences and market dynamics interact to produce resilience outcomes without presupposing that resilience itself is valued. The Pareto frontier between utility and resilience makes these trade-offs explicit and allows evaluation of how much resilience can be maintained for a given subsistence utility level.

Resilient management of ecosystems is not only about the ecosystems themselves, but just as much about the people who inhabit them and depend on them for their livelihoods. In this paper, we contribute to better understanding about the effects of local cultural and economic preferences on the resilience of arid and semi-arid grazing systems, especially in a time where these preferences might be subject to change in a ever globalizing world.

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

References

- Adriansen, H.K., 2005. Pastoral Mobility: A Review. *Nomadic Peoples* 9, 207–214. URL: <https://www.jstor.org/stable/43123755>. publisher: White Horse Press.
- Adriansen, H.K., 2006. Continuity and Change in Pastoral Livelihoods of Senegalese Fulani. *Agriculture and Human Values* 23, 215–229. URL: <https://doi.org/10.1007/s10460-005-6108-3>, doi:doi:10.1007/s10460-005-6108-3.
- Anderies, J.M., Janssen, M.A., Walker, B.H., 2002. Grazing Management, Resilience, and the Dynamics of a Fire-driven Rangeland System. *Ecosystems* 5, 23–44. URL: <https://doi.org/10.1007/s10021-001-0053-9>, doi:doi:10.1007/s10021-001-0053-9.
- Barlow, N., 1987. Harvesting Models for Resource-Limited Populations. *New Zealand Journal of Ecology* 10, 129–133. URL: <https://www.jstor.org/stable/24052795>. publisher: New Zealand Ecological Society.
- Baumgärtner, S., Strunz, S., 2014. The economic insurance value of ecosystem resilience. *Ecological Economics* 101, 21–32. URL: <https://linkinghub.elsevier.com/retrieve/pii/S0921800914000597>, doi:doi:10.1016/j.ecolecon.2014.02.012.

- Berger, J., Buuveibaatar, B., Mishra, C., 2013. Globalization of the Cashmere Market and the Decline of Large Mammals in Central Asia. *Conservation Biology* 27, 679–689. URL: <https://onlinelibrary.wiley.com/doi/abs/10.1111/cobi.12100>, doi:doi:10.1111/cobi.12100. _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1111/cobi.12100>.
- Breman, H., de Wit, C.T., 1983. Rangeland Productivity and Exploitation in Sahel. *Science* 221, 1341–1347.
- Catley, A., Admassu, B., Bekele, G., Abebe, D., 2014. Livestock mortality in pastoralist herds in Ethiopia and implications for drought response. *Disasters* 38, 500–516. doi:doi:10.1111/disa.12060.
- Crépin, A.S., Lindahl, T., 2009. Grazing games: Sharing common property resources with complex dynamics. *Environmental and Resource Economics* 44, 29–46. URL: <https://link.springer.com/article/10.1007/s10640-008-9258-0>, doi:doi:10.1007/S10640-008-9258-0/METRICS.
- Dakos, V., Kéfi, S., 2022. Ecological resilience: what to measure and how. *Environmental Research Letters* 17, 043003. URL: <https://iopscience.iop.org/article/10.1088/1748-9326/ac5767>, doi:doi:10.1088/1748-9326/ac5767.
- Derissen, S., Quaas, M.F., Baumgärtner, S., 2011. The relationship between resilience and sustainability of ecological-economic systems. *Ecological Economics* 70, 1121–1128. URL: <https://linkinghub.elsevier.com/retrieve/pii/S0921800911000103>, doi:doi:10.1016/j.ecolecon.2011.01.003.
- Descroix, L., Luxereau, A., Lambert, L.A., Ruë, O., Diedhiou, A., Diongue-Niang, A., Dia, A.H., Gangneron, F., Manga, S.P., Diedhiou, A.B., Andrieu, J., Chevalier, P., Faty, B., 2024. An Interdisciplinary Approach to Understand the Resilience of Agrosystems in the Sahel and West Africa. *Sustainability* 16, 5555. URL: <https://www.mdpi.com/2071-1050/16/13/5555>, doi:doi:10.3390/su16135555. number: 13 Publisher: Multidisciplinary Digital Publishing Institute.
- Dixon, J., Garrity, D.P., Boffa, J.M., Williams, T.O., Amede, T., Auricht, C., Lott, R., Mburathi, G., 2020. Farming systems and food security in Africa: priorities for science and policy under global

- change. Routledge. URL: <https://hdl.handle.net/10568/109061>, doi:doi:[10.4324/9781315658841](https://doi.org/10.4324/9781315658841).
- D’Odorico, P., Bhattachan, A., Davis, K.F., Ravi, S., Runyan, C.W., 2013. Global desertification: Drivers and feedbacks. *Advances in Water Resources* 51, 326–344. doi:doi:[10.1016/J.ADVWATRES.2012.01.013](https://doi.org/10.1016/J.ADVWATRES.2012.01.013).
- Enfors, E.I., Gordon, L.J., 2007. Analysing resilience in dry-land agro-ecosystems: a case study of the Makanya catchment in Tanzania over the past 50 years. *Land Degradation & Development* 18, 680–696. URL: <https://onlinelibrary.wiley.com/doi/abs/10.1002/ldr.807>, doi:doi:[10.1002/ldr.807](https://doi.org/10.1002/ldr.807). _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1002/ldr.807>.
- Fang, X., Wu, J., 2022. Causes of overgrazing in Inner Mongolian grasslands: Searching for deep leverage points of intervention. *Ecology and Society* 27. URL: <https://ecologyandsociety.org/vol27/iss1/art8/>, doi:doi:[10.5751/ES-12878-270108](https://doi.org/10.5751/ES-12878-270108), publisher: The Resilience Alliance.
- Freier, K.P., Finckh, M., Schneider, U.A., 2014. Adaptation to New Climate by an Old Strategy? Modeling Sedentary and Mobile Pastoralism in Semi-Arid Morocco. *Land* 3, 917–940. URL: <https://www.mdpi.com/2073-445X/3/3/917>, doi:doi:[10.3390/land3030917](https://doi.org/10.3390/land3030917), publisher: Multidisciplinary Digital Publishing Institute.
- Gebremedhn, H.H., Ndiaye, O., Mensah, S., Fassinou, C., Taugourdeau, S., Tagesson, T., Salgado, P., 2023. Grazing effects on vegetation dynamics in the savannah ecosystems of the Sahel. *Ecological Processes* 12, 54. URL: <https://doi.org/10.1186/s13717-023-00468-3>, doi:doi:[10.1186/s13717-023-00468-3](https://doi.org/10.1186/s13717-023-00468-3).
- Gulelat, W., 2002. Household herd size among Pastoralists in relation to overstocking and rangeland degradation. Master’s thesis. International Institute for Geoinformation Science and Earth Observation. Enschede, Netherlands.
- Hauck, S., Rubenstein, D.I., 2017. Pastoralist societies in flux: A conceptual framework analysis of herding and land use among the Mukugodo Maasai of Kenya. *Pastoralism* 7, 18. URL: <https://doi.org/10.1186/s13570-017-0090-4>, doi:doi:[10.1186/s13570-017-0090-4](https://doi.org/10.1186/s13570-017-0090-4).

- Hein, L., Weikard, H.P., 2008. Optimal long-term stocking rates for livestock grazing in a Sahelian rangeland. *African Journal of Agricultural and Resource Economics* 2, 126–150.
- Hein, L.G., Ridder, N.d., 2006. Desertification in the Sahel: a reinterpretation. *Global Change Biology* 12, 751–758. URL: <https://research.wur.nl/en/publications/desertification-in-the-sahel-a-reinterpretation/>, doi:doi:10.1111/j.1365-2486.2006.01135.x, publisher: Wiley.
- Helldén, U., 2008. A coupled human–environment model for desertification simulation and impact studies. *Global and Planetary Change* 64, 158–168. URL: <https://www.sciencedirect.com/science/article/pii/S0921818108001264>, doi:doi:10.1016/j.gloplacha.2008.09.004.
- Heron, R.L., Gertel, J., 2011. *Economic Spaces of Pastoral Production and Commodity Systems : Markets and Livelihoods*. Ashgate Economic Geography Series, Routledge, [N.p.]. URL: <https://research.ebsco.com/linkprocessor/plink?id=1c328188-43a6-3546-bfda-66323e91325d>.
- Higgins, S.I., Kantelhardt, J., Scheiter, S., Boerner, J., 2007. Sustainable management of extensively managed savanna rangelands. *Ecological Economics* 62, 102–114. URL: <https://www.sciencedirect.com/science/article/pii/S092180090600293X>, doi:doi:10.1016/j.ecolecon.2006.05.019.
- Hilker, T., Natsagdorj, E., Waring, R.H., Lyapustin, A., Wang, Y., 2014. Satellite observed widespread decline in Mongolian grasslands largely due to overgrazing. *Global Change Biology* 20, 418–428. URL: <https://onlinelibrary.wiley.com/doi/abs/10.1111/gcb.12365>, doi:doi:10.1111/gcb.12365, _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1111/gcb.12365>.
- Holling, C.S., 1973. Resilience and Stability of Ecological Systems. *Annual Review of Ecology and Systematics* 4, 1–23. URL: <https://www.jstor.org/stable/2096802>, publisher: Annual Reviews.
- Janssen, M.A., Anderies, J.M., Walker, B.H., 2004. Robust strategies for managing rangelands with multiple stable attractors.

- Journal of Environmental Economics and Management 47, 140–162. URL: <https://www.sciencedirect.com/science/article/pii/S009506960300069X>, doi:doi:10.1016/S0095-0696(03)00069-X.
- Kelly, R., Walker, B., 1976. The Effects of Different Forms of Land Use on the Ecology of a Semi-Arid Region in South-Eastern Rhodesia. *Journal of Ecology* 64, 553–576.
- van de Koppel, J., Rietkerk, M., van Langevelde, F., Kumar, L., Klausmeier, C.A., Fryxell, J.M., Hearne, J.W., van Andel, J., de Ridder, N., Skidmore, A., Stroosnijder, L., Prins, H.H.T., 2002. Spatial Heterogeneity and Irreversible Vegetation Change in Semiarid Grazing Systems. *The American Naturalist* 159, 209–218. URL: <https://www.journals.uchicago.edu/doi/10.1086/324791>, doi:doi:10.1086/324791, publisher: The University of Chicago Press.
- Krakovská, H., Kuehn, C., Longo, I.P., 2024. Resilience of dynamical systems. *European Journal of Applied Mathematics* 35, 155–200. URL: https://www.cambridge.org/core/product/identifier/S0956792523000141/type/journal_article, doi:doi:10.1017/S0956792523000141.
- Langelaan, F., van Langevelde, F., Wagener, F., 2026. Stability properties of a dry grazing system. Manuscript in preparation.
- Little, P.D., Debsu, D.N., Tiki, W., 2014. How pastoralists perceive and respond to market opportunities: The case of the Horn of Africa. *Food Policy* 49, 389–397. URL: <https://www.sciencedirect.com/science/article/pii/S0306919214001468>, doi:doi:10.1016/j.foodpol.2014.10.004.
- Mäler, K.G., 2008. Sustainable Development and Resilience in Ecosystems. *Environmental and Resource Economics* 39, 17–24. URL: <http://link.springer.com/10.1007/s10640-007-9175-7>, doi:doi:10.1007/s10640-007-9175-7.
- McCabe, J.T., Quandt, A., Leslie, P.W., 2025. “Those with many cattle were better off”: herd size preferences and perceived tradeoffs in the face of extreme events. *Pastoralism: Research, Policy and Practice* 15, 14511. URL: <https://www.frontierspartnerships.org/journals/>

-
- [pastoralism-research-policy-and-practice/articles/10.3389/past.2025.14511/full](https://doi.org/10.3389/past.2025.14511/full), doi:doi:[10.3389/past.2025.14511](https://doi.org/10.3389/past.2025.14511), publisher: Frontiers Publishing Partnerships.
- Mearns, R., 2004. Sustaining Livelihoods on Mongolia's Pastoral Commons: Insights from a Participatory Poverty Assessment. *Development and Change* 35, 107–139. URL: <https://onlinelibrary.wiley.com/doi/abs/10.1111/j.1467-7660.2004.00345.x>, doi:doi:[10.1111/j.1467-7660.2004.00345.x](https://doi.org/10.1111/j.1467-7660.2004.00345.x), _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1111/j.1467-7660.2004.00345.x>.
- Moritz, M., Buffington, A., Yoak, A.J., Hamilton, I.M., Garabed, R., 2017. No Magic Number: an Examination of the Herd-Size Threshold in Pastoral Systems Using Agent-Based Modeling. *Human Ecology* 45, 525–532. URL: <http://link.springer.com/10.1007/s10745-017-9927-0>, doi:doi:[10.1007/s10745-017-9927-0](https://doi.org/10.1007/s10745-017-9927-0).
- Nyariki, D.M., Amwata, D.A., 2019. The value of pastoralism in Kenya: Application of total economic value approach. *Pastoralism* 9, 9. URL: <https://doi.org/10.1186/s13570-019-0144-x>, doi:doi:[10.1186/s13570-019-0144-x](https://doi.org/10.1186/s13570-019-0144-x).
- OECD, 2025. Unseen markets: A data analysis of intra-regional food trade in West Africa: Intra-regional Food Trade in West Africa. URL: https://www.oecd.org/en/publications/intra-regional-food-trade-in-west-africa_3f5fd54a-en/full-report/component-6.html.
- Ouédraogo, K., Zaré, A., Korbéogo, G., Ouédraogo, O., Linstädter, A., 2021. Resilience strategies of West African pastoralists in response to scarce forage resources. *Pastoralism* 11, 16. URL: <https://doi.org/10.1186/s13570-021-00210-8>, doi:doi:[10.1186/s13570-021-00210-8](https://doi.org/10.1186/s13570-021-00210-8).
- Perrings, C., Walker, B., 2004. Conservation in the optimal use of rangelands. *Ecological Economics* 49, 119–128.
- Quaas, M.F., Baumgärtner, S., Becker, C., Frank, K., Müller, B., 2007. Uncertainty and sustainability in the management of rangelands. *Ecological Economics* 62, 251–266. URL: <https://doi.org/10.1016/j.ecolecon.2007.05.001>.

-
- [//linkinghub.elsevier.com/retrieve/pii/S0921800906002667](https://linkinghub.elsevier.com/retrieve/pii/S0921800906002667),
doi:doi:[10.1016/j.ecolecon.2006.03.028](https://doi.org/10.1016/j.ecolecon.2006.03.028).
- Rahimi, J., Haas, E., Grote, R., Kraus, D., Smerald, A., Laux, P., Goopy, J., Butterbach-Bahl, K., 2021. Beyond livestock carrying capacity in the Sahelian and Sudanian zones of West Africa. *Scientific Reports* 11, 22094. URL: <https://www.nature.com/articles/s41598-021-01706-4>, doi:doi:[10.1038/s41598-021-01706-4](https://doi.org/10.1038/s41598-021-01706-4).
- Rietkerk, M., Bastiaansen, R., Banerjee, S., van de Koppel, J., Baudena, M., Doelman, A., 2021. Evasion of tipping in complex systems through spatial pattern formation. *Science* 374, eabj0359. URL: <https://www.science.org/doi/10.1126/science.abj0359>, doi:doi:[10.1126/science.abj0359](https://doi.org/10.1126/science.abj0359). publisher: American Association for the Advancement of Science.
- Rietkerk, M., Van Den Bosch, F., Van De Koppel, J., 1997a. Site-Specific Properties and Irreversible Vegetation Changes in Semi-Arid Grazing Systems. *Oikos* 80, 241–252.
- Rietkerk, M., Van Den Bosch, F., Van De Koppel, J., 1997b. Site-Specific Properties and Irreversible Vegetation Changes in Semi-Arid Grazing Systems. *Oikos* 80, 241. URL: <https://www.jstor.org/stable/3546592?origin=crossref>, doi:doi:[10.2307/3546592](https://doi.org/10.2307/3546592).
- Short, J., 1985. The Functional Response of Kangaroos, Sheep and Rabbits in an Arid Grazing System. *Journal of Applied Ecology* 22, 435–447. URL: <https://www.jstor.org/stable/2403176>, doi:doi:[10.2307/2403176](https://doi.org/10.2307/2403176). publisher: [British Ecological Society, Wiley].
- Stigter, J., Van Langevelde, F., 2004. Optimal harvesting in a two-species model under critical depensation. *Ecological Modelling* 179, 153–161. URL: <https://linkinghub.elsevier.com/retrieve/pii/S0304380004002534>, doi:doi:[10.1016/j.ecolmodel.2004.06.003](https://doi.org/10.1016/j.ecolmodel.2004.06.003).
- Turner, M.D., McPeak, J.G., Ayantunde, A., 2014. The Role of Livestock Mobility in the Livelihood Strategies of Rural Peoples in Semi-Arid West Africa. *Human Ecology* 42, 231–247. URL: <https://doi.org/10.1007/s10745-013-9636-2>, doi:doi:[10.1007/s10745-013-9636-2](https://doi.org/10.1007/s10745-013-9636-2).

- Turner, M.D., Schlecht, E., 2019. Livestock mobility in sub-Saharan Africa: A critical review. *Pastoralism* 9, 13. URL: <https://doi.org/10.1186/s13570-019-0150-z>, doi:doi:10.1186/s13570-019-0150-z.
- Turner, M.D., Williams, T.O., 2002. Livestock Market Dynamics and Local Vulnerabilities in the Sahel. *World Development* 30, 683–705. URL: <https://www.sciencedirect.com/science/article/pii/S0305750X01001334>, doi:doi:10.1016/S0305-750X(01)00133-4.
- van de Koppel, J., Rietkerk, M., 2000. Herbivore regulation and irreversible vegetation change in semi-arid grazing systems. *Oikos* 90, 253–260.
- van de Koppel, J., Rietkerk, M., Weissing, F., 1997. Catastrophic vegetation shifts and soil degradation in terrestrial grazing systems. *Trends in Ecology & Evolution* 12, 352–356.
- Vetter, S., 2005. Rangelands at equilibrium and non-equilibrium: recent developments in the debate. *Journal of Arid Environments* 62, 321–341. URL: <https://www.sciencedirect.com/science/article/pii/S014019630500011X>, doi:doi:10.1016/j.jaridenv.2004.11.015.
- Vignal, T., Baudena, M., Mayor, A.G., Sherratt, J.A., 2023. Impact of different destocking strategies on the resilience of dry rangelands. *Ecology and Evolution* 13, e10102. URL: <https://onlinelibrary.wiley.com/doi/abs/10.1002/ece3.10102>, doi:doi:10.1002/ece3.10102, _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1002/ece3.10102>.
- Walker, B., Ludwig, D., Holling, C., Peterman, R., 1981. Stability of semi-arid savanna grazing systems. *Journal of Ecology* 69, 473–498.
- Weikard, H.P., Hein, L., 2011. Efficient versus Sustainable Livestock Grazing in the Sahel. *Journal of Agricultural Economics* 62, 153–171. URL: <https://onlinelibrary.wiley.com/doi/full/10.1111/j.1477-9552.2010.00275.x>, doi:doi:10.1111/J.1477-9552.2010.00275.X.
- Wendling, V., Peugeot, C., Mayor, A.G., Hiernaux, P., Mougin, E., Grippa, M., Kergoat, L., Walcker, R., Galle, S., Lebel, T., 2019. Drought-induced regime shift and resilience of a Sahelian ecohydrosystem. *Environmental Research Letters* 14, 105005. URL: <https://dx.doi.org/10.1088/1748-9326/ab3dde>, doi:doi:10.1088/1748-9326/ab3dde, publisher: IOP Publishing.

- van Wijngaarden, W., 1985. Elephants - Trees- Grass - Grazers: Relationships Between Climate, Soils, Vegetation and Large Herbivores in a Semi-Arid Savanna Ecosystem (Tsavo, Kenya) - ProQuest. Ph.D. thesis. Wageningen University and Research. URL: <https://www.proquest.com/openview/6c34cd3ed0a91cacea65dc813d16ac40/1?pq-origsite=gscholar&cbl=2026366&diss=y>.
- Yatat-Djeumen, I.V., Doyen, L., Tewa, J.J., Ghosh, B., 2025. Bioeconomic Viability and Resilience of Savanna. *Environmental Modeling & Assessment* 30, 71–85. URL: <https://doi.org/10.1007/s10666-024-09989-3>, doi:doi:[10.1007/s10666-024-09989-3](https://doi.org/10.1007/s10666-024-09989-3).
- Zhao, H.L., Zhao, X.Y., Zhou, R.L., Zhang, T.H., Drake, S., 2005. Desertification processes due to heavy grazing in sandy rangeland, Inner Mongolia. *Journal of Arid Environments* 62, 309–319. URL: <https://www.sciencedirect.com/science/article/pii/S0140196305000054>, doi:doi:[10.1016/j.jaridenv.2004.11.009](https://doi.org/10.1016/j.jaridenv.2004.11.009).