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Effects of different management regimes on soil erosion and surface runoff in semi-arid to sub-humid rangelands

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A B S T R A C T

Over one billion people's livelihoods depend on dry rangelands through livestock grazing and agriculture. Livestock grazing and other management activities can cause soil erosion, increase surface runoff and reduce water availability. We studied the effects of different management regimes on soil erosion and surface runoff in semi-arid to sub-humid rangelands. Eleven management regimes were assessed, which reflected different livestock grazing intensities and rangeland conservation strategies. Our review yielded key indicators for quantifying soil erosion and surface runoff. The values of these indicators were compared between management regimes. Mean annual soil loss values in the '*natural ungrazed*', '*low intensity grazed*', '*high intensity grazed rangelands*' and '*man-made pastures*' regimes were, respectively, 717 (SE = 388), 1370 (648), 4048 (1517) and 4249 (1529) kg ha^{-1} yr^{-1}. Mean surface runoff values for the same regimes were 98 (42), 170 (43), 505 (113) and 919 (267) m^3 ha^{-1} yr^{-1}, respectively. Soil loss and runoff decreased with decreasing canopy cover and increased with increasing slope. Further analyses suggest that livestock grazing abandonment and '*exotic plantations*' reduce soil loss and runoff. Our findings show that soil erosion and surface runoff differ per management regime, and that conserving and restoring vulnerable semi-arid and sub-humid rangelands can reduce the risks of degradation.

1. Introduction

Drylands cover about 41% of the Earth's land surface and are inhabited by more than two billion people, of which 90% live in developing countries (UN, 2011). Over one billion people in these areas depend directly on drylands for their livelihoods, mostly through livestock grazing (65%) and agriculture (25%) (MA, 2005; UN, 2011). Although livestock grazing in drylands contributes to less than 20% of the global meat and dairy production, half of the world's livestock is supported by drylands' natural productivity (MA, 2005). The aridity index (AI) (i.e. the ratio between annual precipitation (P) and annual potential evapotranspiration (PET)) characterizes drylands, which occur in areas where AI $\leq$ 0.65 (i.e. PET is at least 50% larger than P) (Middleton and Thomas, 1997). Drylands are thus limited by soil moisture resulting from low rainfall and high evaporation.

Twelve to seventeen dryland major types are distinguished, aggregated into four 'broad' biomes: desert, grassland, Mediterranean scrubland, and dry woodlands (MA, 2005). These biomes largely follow the aridity gradient: AIs of hyper-arid, arid, semi-arid and sub-humid drylands range, respectively, from less than 0.05, 0.05 to 0.2, 0.2 to 0.5 and 0.5 to 0.65 (Middleton and Thomas, 1997). In this study we focus on semi-arid and sub-humid drylands and will refer to them as 'rangelands', unless specified differently.

Land degradation is a common threat to semi-arid and sub-humid rangelands. Population increase, climatic variations and human activities (i.e. management) drive land degradation (MA, 2005; UN, 2011). Degradation refers to reduced or lost biological or economic productivity and complexity of both natural and managed rangelands (MA, 2005). Approximately one fifth of all rangelands are currently suffering from degradation (MA, 2005). Rangelands are dominated by grasses, forbs, shrubs and dispersed trees (Westoby et al., 1989). Rangelands are often associated with grazing and managed by ecological or low intensity management

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(Jouven et al., 2010). Most rangelands are grazed by livestock but some rangelands are grazed by natural grazers (Jouven et al., 2010). Semi-arid and sub-humid rangelands cover 56 million km^2 globally (UN, 2011) and are sensitive to management effects and climate variability. Sub-humid rangelands are, due to their higher water availability, increasingly used for intensive livestock grazing and cropping. Semi-arid rangelands, especially in the Mediterranean, have been grazed since the late 1900s (Perevolotsky and Seligman, 1998). This relatively 'recent' disturbance has resulted in a transition from grass-dominated to shrub-dominated rangelands and has led to increased rain-induced soil erosion and increased surface runoff (Perevolotsky and Seligman, 1998; Stringham et al., 2003).

The effects of rangeland management and land-use change on degradation and agricultural productivity are poorly understood (UN, 2011). Preventing soil erosion and runoff is crucial to reverse degradation and improve productivity. Reports by MA (2005) and TEEB (2010) acknowledged this by including soil erosion prevention and water flow regulation as important ecosystem services (i.e. the contributions to human wellbeing). Both ecosystem services depend on similar underlying ecological characteristics (Fu et al., 2011). Soil erosion prevention reduces loss of productive land, downstream water pollution, clogging of waterways, flood risk and improves productivity (Snyman, 1999; Fu et al., 2011). Reducing surface runoff provides similar benefits, as well as constant water availability to vegetation, decreased sedimentation and nutrient loss (Narain et al., 1997; van Luijk et al., 2013). Rangeland management is a crucial factor to consider because it negatively or positively affects soil erosion and runoff.

This study assessed the consequences of management decisions in semi-arid and sub-humid rangelands by studying the effects of different management regimes on soil erosion and surface runoff. Management regimes are 'the bundle of human activities that serve land-use purposes' and reflect different land-use intensities. Despite the vast scientific consensus on the impacts of different livestock grazing intensities, many vaguely defined and even subjective categories can be found in the rangeland literature, ranging from 'proper' to 'somewhat overgrazed' (Smith, 1940), 'moderate' (Snyman, 1998), 'heavy' and 'very heavy' (Dormaar et al., 1994; Mwendera and Saleem, 1997). We developed a comprehensive typology of management regimes in semi-arid and sub-humid rangelands, based on eight qualitative management indicators. We also identified indicators for quantifying soil erosion and runoff, based on a targeted review of peer-reviewed papers. Quantifications from these studies were then used to establish mean values of soil erosion and runoff related to different management regimes. In our analysis, we focused on regimes differing in livestock grazing intensities, as well as rangeland restoration and conservation. By comparing different management regimes we identified regimes with the least erosion and optimal runoff and, thus, quantified the related ecosystem services.

2. Methods

2.1. Indicator selection for quantifying soil erosion and surface runoff

We consulted well-cited review papers on soil erosion and/or surface runoff, which resulted in an overview of recurring indicators to quantify soil erosion and runoff indicators for soil erosion and runoff. These papers were by Kosmas et al. (1997), Cantón et al. (2001), Fu et al. (2009), García-Ruiz (2010) and Fu et al. (2011). We then consulted publications that were either citing or cited by these five review papers, thereby focussing on papers that quantified livestock and rangeland restoration management effects on soil erosion and/or runoff. Only indicators that recurred in the literature were included in our overview of indicators, which is provided in Section 4 (including further references) and formed the basis for the analysis that is described in Section 2.3.

2.2. Developing a management regime typology

Our management regime typology included five broad categories, based on Alkemade et al. (2013): 'natural', 'low intensity use', 'high intensity use', 'converted' and 'abandoned'. Each management regime should consist of distinguishable land-use activities, and resulting land cover and specific ecological and socio-economic characteristics. Land use is the purpose for which humans change land cover to their own benefit (Verburg et al., 2011). Land use is enabled by a series of activities, which comprise the management regime (Van Oudenhoven et al., 2012). Land cover refers to all physical biotic and abiotic components that make up landscapes, including vegetation, soils, cropland, water and human structures (Verburg et al., 2011). Moreover, management regimes are assumed to be hard to reverse and transitions from one regime to another require substantial time, investments and management actions (Westoby et al., 1989). With this in mind, further information was needed to select indicators that would help distinguishing different regimes.

A targeted literature review on Web of Science™ was conducted, using the keywords 'semi-arid' OR '*sub-humid* OR 'dryland' combined with '*grazing' OR 'livestock' OR 'rangeland' OR 'land use' OR 'ecology' OR '*degradation' OR '*management'. Papers were selected from the top-50 most relevant search returns and checked for management indicators and potential management regime categories. We only considered studies dealing with livestock grazing and nature conservation (e.g. restoration, protection, abandoning grazing, reversing erosion) in rangelands and converted rangelands in semi-arid or sub-humid areas. However, an additional search was required to identify indicators for *overgrazed, abandoned rangelands and silvo-pasture*. The studies' aridity zone was verified using a 10' 'Global map of aridity' (FAO, 2014). Locations were approximated when limited information was provided. When aridity zones mentioned in the study's site description did not match ours, we used the studies' original description if the study sites were located between two aridity zones or if the study was conducted before 1990. We ignored the study's description if the site was located more than 300 km away from the claimed aridity zone. We reviewed suitable studies to find indicators for different management regimes, which are summarised in Table 1.

Several assumptions were made to make the indicators applicable to a large variety of rangeland ecosystems and to cope with different ways how rangeland management and land use are described. No quantitative ranges were determined for stocking rate intensities, because these depend on local factors that differ throughout the world's rangelands. Intermediate classes between low and high, and high and overgrazing were ill-defined and highly variable, and thus not considered. Many studies also report the 'rangeland condition' and/or vegetation cover in response to different intensities of grazing without referring to actual stocking rates. These indicators are frequently used in traditional rangeland ecology studies (e.g. Snyman, 1997; Puttick et al., 2011). Rangeland condition and/or vegetation cover approximate stocking rates. For instance, poor, good and degraded rangelands could generally be linked to low, high and overgrazed stocking rates, respectively (Snyman, 1997; Fynn and O'Connor, 2000). Rangeland condition involves ecological status (i.e. botanical composition and cover, and plant successional status, productivity, nutritive value and palatability) (Snyman, 1999). Water use efficiency, above-ground

Table 1
Indicators used for developing management regimes in semi-arid and sub-humid rangelands.

Indicator	Short description *Categories plus abbreviations*	Sources
Stocking rate	Stocking rate relative to rangeland's grazing capacity. Low is much below the grazing capacity (<50%), high is around grazing capacity and overgrazed is much above grazing capacity. Categories: *Low (L), High (H), Overgrazed (O)*	McIvor et al. (1995); Mwendera and Saleem (1997); Dormaar and Willms (1998); Oztas et al. (2003)
Rangeland condition	Rangeland's ecological status (botanical composition and cover, and plant successional state), and its productivity, nutritive value and palatability. Categories: *Poor (P), Good (G), Degraded (D)*	Snyman (1999); Fynn and O'Connor (2000); Lechmere-Oertel (2003)
Vegetation cover	Vegetation cover in response to different intensities of livestock grazing. This indicator only features general descriptions of vegetation cover, such as mature vegetation, grass, invasive woody species, bare soil etc.	Westoby et al. (1989); Stringham et al. (2003); Puttick et al. (2011); Manjoro et al. (2012)
Exclosing or enclosing	Exclosing involves disabling livestock grazing with fences and enclosing enables more localised grazing. Fences or natural barriers are used. Categories: *Exclosing (Ex), Enclosing (En)*	Launchbaugh (1955); Dormaar and Willms (1998); Reeder et al. (2004); de Aguiar et al. (2010)
Intercropping	The occurrence of trees combined with grazing. Trees can be natural or planted and grazing land can include rangelands or sown pastures. Categories: *Yes (Y), No (N)*	McIvor et al. (1995); Narain et al. (1997); de Aguiar et al. (2010)
Soil treatment	Treating the topsoil layer to optimise livestock grazing. Examples include removing topsoil, removing weeds, ploughing and mulching. Categories: *Yes (Y), No (N)*	Simanton et al. (1985); McIvor et al. (1995); Mwendera and Saleem (1997)
Vegetation removal	Removing unwanted vegetation, that inhibits grass production. Examples include woody, unpalatable and water-consuming species. Categories: *Yes (Y), No (N)*	Simanton et al. (1985); Dormaar et al. (1994); McIvor et al. (1995); de Aguiar et al. (2010)
Restoring or planting	Planting exotic (Exot) or natural (Nat) vegetation to reduce erosion, increase wood production etc.	Narain et al. (1997); Andreu et al. (1998); Lechmere-Oertel (2003)
Sowing grass	Sowing nutritious species on pre-treated and cleared soils or natural rangelands with the aim to maximise nutrient intake of livestock. Categories:*Yes (Y), No (N)*	Dormaar et al. (1994); McIvor et al. (1995); Narain et al. (1997);
Fertilizers, pesticides, herbicides use	Applying fertilizers (F), herbicides (H) and/or pesticides (P) to improve grass productivity.	Narain et al. (1997); de Aguiar et al. (2010)

production and basal cover generally decrease when rangelands degrade and the vegetation deteriorates (Snyman, 1999; Puttick et al., 2011). Vegetation cover, here a qualitative indicator, is used to describe distinctive management-induced ecological states (Westoby et al., 1989). Although semi-arid and sub-humid rangelands comprise many different ecosystems, we used the rangeland ecology literature (as listed in Table 1) to confirm the rangelands' management-induced vegetation cover.

For the ‘exclosing/enclosing livestock’ indicator we assumed that roaming wildlife would be excluded. Exclosure or enclosure's duration was only considered when distinguishing between abandoned and conservation rangelands. The indicator ‘intercropping’ applies to both sown pasture systems and natural rangelands. We considered intercropping when trees were combined with high stocking rates. Low stocking rates generally exclude enclosing grazing areas and thus by default occur on rangelands were trees could occur naturally. Finally, most studies mention fertilizers, pesticides or herbicides only as a pre-treatment prior to experiments or new grazing regimes. Very few studies mention them as current activities, except for pasture management.

We created eleven different management regimes based on the indicators in Table 1 and the literature's descriptions of ‘management regimes’. We cross-tabulated the regimes and indicators to develop the typology, which is provided in Section 3. Each management regime is characterised by at least two differing indicators from other regimes.

2.3. Quantifying soil erosion and runoff per management regime

We used Web of Science™ and Google Scholar™, and used the management regimes' names or synonyms as keywords combined with *erosion* and *runoff*. Search results were sorted by ‘relevance’ and the top-50 papers were selected and checked for quantitative data. This search yielded 26 papers that corresponded with our criteria, with 267 data entries from 18 studies on soil erosion measurements and 283 data entries from 25 studies on surface runoff measurements. Fig. 1 shows the studies' reported locations reported in the papers and Table A1 (Appendix A) lists all studies that were included.

Graphical data, if required and useful, were extracted using Plot Digitizer (version 2.6.3). All data was stored. Data that was linked to multiple (underpinning) indicators was entered as one data entry. Apart from quantified indicators, we also registered location (coordinates, country, location description), measurement date and annual precipitation. We assigned a management regime to each data entry based on matching indicators mentioned in the study. We used ‘stocking rate’ as a guiding indicator and rangeland condition or vegetation cover to either verify the mentioned stocking rate, or to indicate the stocking rate indirectly. However, qualitative terms for stocking rate, such as ‘high’, ‘heavy’, ‘low’, ‘overgrazed’ and ‘degraded’, were used inconsistently by the authors. We only took a given stocking rate when it was related to the rangeland's grazing or carrying capacity (c.f. Table 1). Quantitative stocking rates were sometimes used to compare intensities within studies. Data entries that could not be linked to management regimes (e.g. bare soil and cropland measurements), were excluded from our analysis.

Mean values were calculated per management regime for all quantitative indicators as listed in Section 4. Differences between means were not statistically tested, due to large differences in number of studies and data entries per management regime. We conducted a Spearman's rank order correlation between the underlying quantified indicators and soil loss and runoff data. This analysis was meant to identify key indicators that might explain differences in soil loss and runoff and was conducted for all data together (i.e. not separated per management regime).

Further analyses were performed to quantify ecosystem service provision (soil loss prevention and water flow regulation) involved in transitions between management regimes. Mean values for soil loss and surface runoff were obtained per study and differences

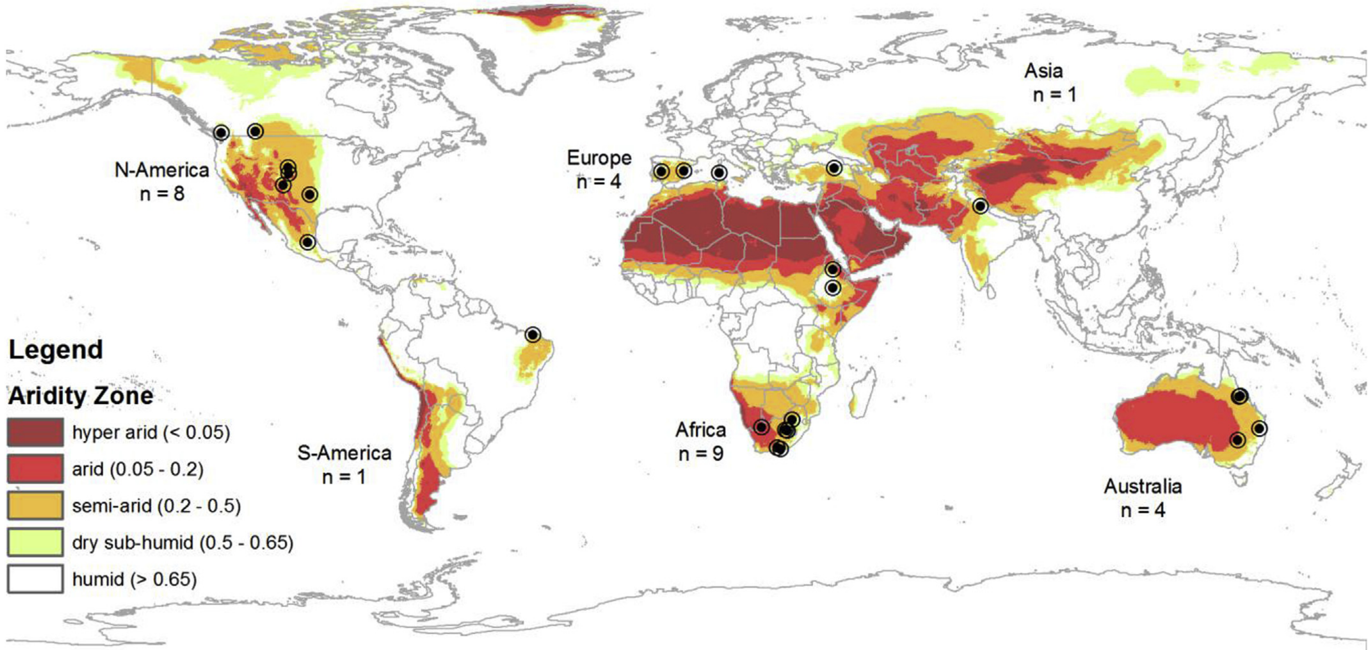

Fig. 1. Analysed studies' reported locations on the on the Global Aridity Map (FAO, 2014).

between means were subjected to a T-Test ($p < 0.05$). Mean values were averaged when multiple studies had quantified the same management regimes. We used eight out of thirteen studies on soil loss (103 data entries) and three out of eleven on surface runoff (54 data entries) for this further analysis. Data were analysed with SPSS Statistics version 22.

3. Management regime typology

We derived eleven management regimes, divided into 5 broad categories, for semi-arid and sub-humid rangelands and considered livestock grazing and nature conservation as main land-use purposes. The five broad categories of management regimes (Table 2) are defined as follows:

- *Natural rangelands* are not grazed by livestock but can be grazed by natural grazers and have a recognised high biodiversity value or ecological function. Management activities are limited to nature protection (fences, patrolling etc.) but vegetation and soils are undisturbed. The category includes *natural ungrazed* (e.g. Launchbaugh, 1955; Andreu et al., 1998) and *conservation rangelands* (e.g. Reeder et al., 2004; van Luijk et al., 2013).
- *Low intensity use rangelands* can be natural or restored, and are managed to support either low intensity livestock grazing or natural vegetation recovery. Management activities do not involve infrastructure construction, but may modify vegetation cover. These rangelands include *low intensity grazed rangelands* (e.g. McIvor et al., 1995) and *restoration rangelands* (e.g. Andreu et al., 1998).
- *High intensity use rangelands* are managed for maximised livestock grazing. Management activities include high stocking rates, introducing highly palatable grass species, pastures intercropped with trees, and using chemical inputs to optimise grass productivity. The category includes *high intensity grazed rangelands* (e.g. Mwendera and Saleem, 1997), *overgrazed rangelands* (e.g. Oztas et al., 2003) and *silvo-pastures* (e.g. McIvor et al., 1995; de Aguiar et al., 2010).
- *Converted rangelands* are systems in which the original vegetation has been cleared and replaced to serve another land-use purpose, such as livestock grazing or reduce erosion resulting from overgrazing. Management activities can be sowing grass, planting trees, irrigation, applying pesticides and herbicides, and ploughing. This category includes *man-made pastures* (e.g. McIvor et al., 1995) and *exotic tree plantations*, which are planted to reduce grazing-induced erosion (e.g. Narain et al., 1997).
- *Abandoned rangelands and pastures* have been used intensively or unsustainably, and are currently without a land-use purpose. This category includes both recovering *abandoned rangelands* (e.g. Descheemaeker et al., 2006) and irreversibly *degraded abandoned rangelands* (e.g. Muñoz-Robles et al., 2011).

Most management regimes are characterised by distinctively different stocking rates or livestock management (Table 2). Moreover, increasing stocking rates coincide with increasing additional management efforts and inputs, such as vegetation removal, soil treatment and applying herbicides. Along the typology, vegetation cover changes from mature vegetation (*natural rangelands*) to more grassy species (*low intensity grazed*) and introduced grass and herbaceous species (*high intensity grazed*). *Overgrazed rangelands* are characterised by bare soils and woody encroachment, whereas *abandoned degraded rangelands* are suffering from desertification and increased woody encroachment (Puttick et al., 2011; Manjoro et al., 2012). Rangeland degradation involves irreversible changes in both soils and vegetation (Fynn and O'Connor, 2000).

Several other management regimes aim to restore rangelands' productivity and/or original biodiversity and additional indicators are needed to distinguish regimes that aim to reverse land degradation and restore or conserve rangelands. For instance, *conservation* and *restoration rangelands* are both 'exclosed', but active restoration management (i.e. replanting, removing alien vegetation etc.) only takes place in the latter. In fact, *conservation rangelands* could be described as 'undergrazed', since neither livestock nor free-roaming wildlife grazes here. *Silvo-pastures* aim to increase the rangelands' productivity through intercropping with trees. *Exotic tree plantations* only aim to reverse soil erosion but could actually

Table 2
Short description and management indicators of management regimes in semi-arid and sub-humid rangelands. Management indicators are further explained in Table 1, acronyms are repeated below.

	Short description	Stocking rate (L, H, O)	Ex- or en-closing (Ex, En)	Inter-cropping (Y/N)	Soil treatment (Y/N)	Vegetation removal (Y/N)	Restoring (Nat), planting (Exot)	Sowing grass (Y/N)	Using F, P, H
I. Natural									
Natural ungrazed rangeland	Grazed only by free-roaming natural grazers. Good rangeland condition; undisturbed mature vegetation.	–	–	N	N	N	–	N	–
Conservation rangeland	All grazing disabled for >40 years, to optimize vegetation recovery. Good rangeland condition.	–	Ex	N	N	N	Nat	N	–
II. Low intensity use									
Low intensity grazed rangeland	Livestock grazing below the carrying capacity. Some palatable grasses persist. Good rangeland condition	L	–	Y	N	N	–	N	–
Restoration rangeland	Actively restored former grazing land.	–	Ex	N	N	Y	Nat	N	–
III. High Intensity use									
High intensity grazed rangeland	Livestock grazing at carrying capacity. Altered vegetation and soils. Poor rangeland condition.	H	En	N	Y	Y	–	Y	F,H
Overgrazed rangeland	Continuous grazing above carrying capacity. Degraded condition; woody encroachment and some bare soils.	O	En	N	Y	Y	–	N	F
Silvo-pasture	Rangelands or sown pastures intercropped with trees to provide shade, fodder or to prevent erosion.	L/H	En/Ex	Y	Y	Y/N	Nat/Exot	Y	P,H
IV. Converted									
Man-made pasture	Original vegetation cleared and replaced by optimal grass for livestock grazing on pre-treated soils.	L/H	En	N	Y	Y	–	Y	F,P,H
Exotic tree plantation	Exotic trees planted on formerly degraded land, with the aim to reduce soil erosion resulting from grazing.	–	Ex	N	Y	Y	Exot	N	P,H
V. Abandoned									
Abandoned rangeland	Rangelands or pastures relieved from grazing for <30 years, allowing the vegetation to recover.	–	Ex	N	N	Y	–	N	–
Abandoned degraded rangeland	No longer grazed or used due to irreversible changes in soils (bare) and vegetation (woody encroachment).	–	–	N	N	–	–	N	–

Note: Dashes (–) indicate when information could not be found or indicators do not apply. Acronyms: Low (L), High (H), Overgrazed (O), Yes (Y), No (N), Natural vegetation (Nat), Exotic vegetation (Exot), Fertilizer (F), Pesticide (P), Herbicide (H).

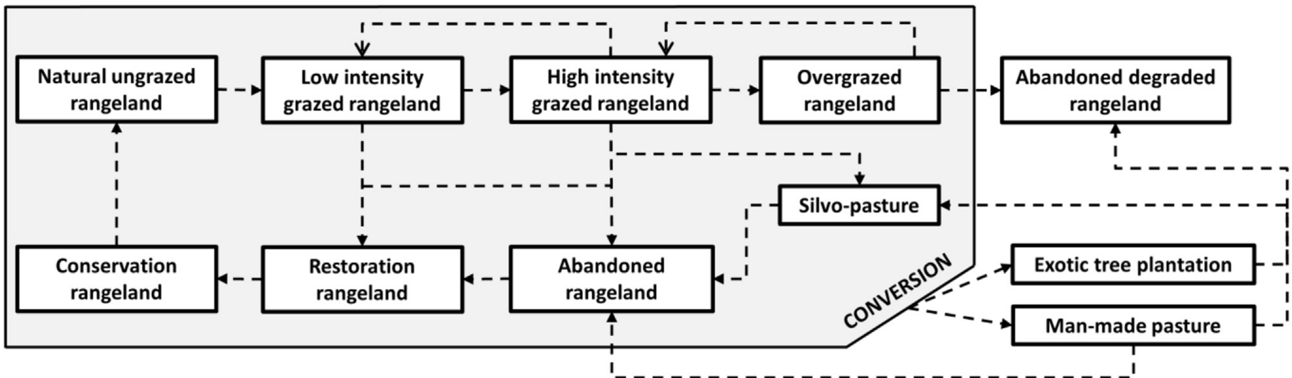

Fig. 2. Possible transitions between management regimes of semi-arid and sub-humid rangelands. Based on Bellamy and Brown (1994), Stringham et al. (2003) and Alkemade et al. (2013).

have other negative effects on rangelands' productivity and biodiversity. *Abandoned rangelands*, finally, are characterised by a shorter period of not grazing and could either be used for livestock grazing or continue to become conservation rangelands.

Possible transitions between management regimes are illustrated in Fig. 2. These transitions include land-use intensification, restoration or reducing land-use intensity. Each transition requires additional and prolonged management activities (Stringham et al., 2003). All regimes could lead to conversion into *exotic tree plantations* or *man-made pastures*. *Abandoned degraded rangelands* are mostly 'end of line' management regimes, where stepwise restoration is impossible. Fig. 2 will be used in Section 5.3 to illustrate differences in soil erosion and surface runoff between management regimes. The figure helps to inform decision makers because all management regimes and transitions between them represent clear management choices.

4. Indicators for quantifying soil erosion and surface runoff

Recurring indicators from the reviewed literature for soil erosion and surface runoff are provided in Fig. 3 and listed in Table B1 (Appendix B). Most studies (seventeen) assessed both erosion and runoff. Fourteen studies assessed only soil erosion and eleven assessed only runoff, respectively. Studies that assess both erosion and runoff use more recurring indicators on a consistent basis. Moreover, studies on just erosion or runoff used indicators that were rarely used by other studies. We note, for example, that annual soil loss was only measured by two studies that focused on erosion only, as compared to eleven that assessed both services. Similarly, annual surface runoff was only measured by three studies, as compared to eight that assessed both services. A possible explanation could be that mono-disciplinary studies generally follow a more detailed research approach. We collected twelve underlying indicators for both soil erosion and runoff. Key indicators are described below.

The relationships between the indicators are illustrated in an indicator interaction diagram (Fig. 3). This diagram depicts information flows rather than matter flows, and connects indicators rather than processes or systemic components. Therefore, the interaction diagram does not explain the dynamic complexity of soil erosion and runoff. 'Soil loss' per area (and per year) approximates soil erosion and 'surface runoff' per area (and per year) approximates surface runoff (e.g. Narain et al., 1997; Cantón et al., 2001; Fu et al., 2011). Most values for 'annual soil loss' and 'annual surface runoff' include all measured rain events of a given year or averaged values of multiple years. Some studies do not specify whether they assessed surface runoff or total runoff (i.e. including sub-surface runoff and drainage). For most studies, we could establish what was measured from the experimental setup, but 'soil loss' and 'surface runoff' only measured per unit of area, lacks the temporal dimension. Standardizing and comparing data is therefore difficult. Some studies measured individual rain events, while others focused on seasons, years or longer periods. Finally, hydrological studies often measure runoff as a percentage of total rainfall (i.e. the 'surface runoff coefficient'), which can also be daily, seasonal or annual rainfall. The indicators underpinning soil loss and surface runoff can be grouped into four categories: rainfall, vegetation, and topography and soil (Fig. 3).

Rainfall is positively correlated with soil loss and surface runoff (Kosmas et al., 1997; Vásquez-Méndez et al., 2010). Although 'annual rainfall' (m^3 ha^{-1} yr^{-1}) can be useful to relate annual soil loss or surface runoff (Le Maitre et al., 1999), most studies provide rainfall data rather than measuring and incorporating soil loss or runoff explicitly. More common indicators are 'total rainfall' (m^3 ha^{-1}) resulting from hydrological experiments, or 'rainfall erosive events' (m^3 ha^{-1}) resulting from erosion experiments. Both indicators approximate the amount of rainfall during a given observation period (Bartley et al., 2006). 'Rainfall intensity' indicates hourly rainfall per area and informs on soil loss and runoff during high and low intensity rainfall events. This indicator is used infrequently because erosion and runoff events are usually assessed over longer periods. Most studies also describe the period during which most rainfall occurs or was measured (i.e. the 'rainfall regime') enabling to identify peak runoffs (Cerda et al., 1998). The amount of rainfall not intercepted by vegetation (i.e. 'throughfall'), largely determines the amount of surface runoff (Mills and Fey, 2004). Most studies mentioned only throughfall, but 'interceptional loss' (i.e. rainfall minus throughfall) could be calculated. Interception rates are established for different vegetation types (e.g. Dunkerley, 2000). 'Evapotranspiration' describes the amount of rainfall that is returned directly to the atmosphere by transpiration (from plants) and evaporation (from soils) (Cantón et al., 2001). Most studies, however, establish evapotranspiration indirectly without relating it to surface runoff.

'Vegetation cover' (i.e. % of total land cover) is negatively correlated to both soil loss and surface runoff (Vásquez-Méndez et al., 2010; Fu et al., 2011). Vegetation intercepts raindrops, reduces raindrop impacts and promotes infiltration pathways (Le Maitre et al., 1999). Land management activities and ecosystem and vegetation type determine vegetation cover's structure and density (e.g. Le Maitre et al., 1999; van Luijk et al., 2013). For instance, plant communities in semi-arid scrublands take up water more efficiently than plants in sub-humid ecosystems (Le Maitre et al., 1999; Mills and Fey, 2004). Lower vegetation cover in

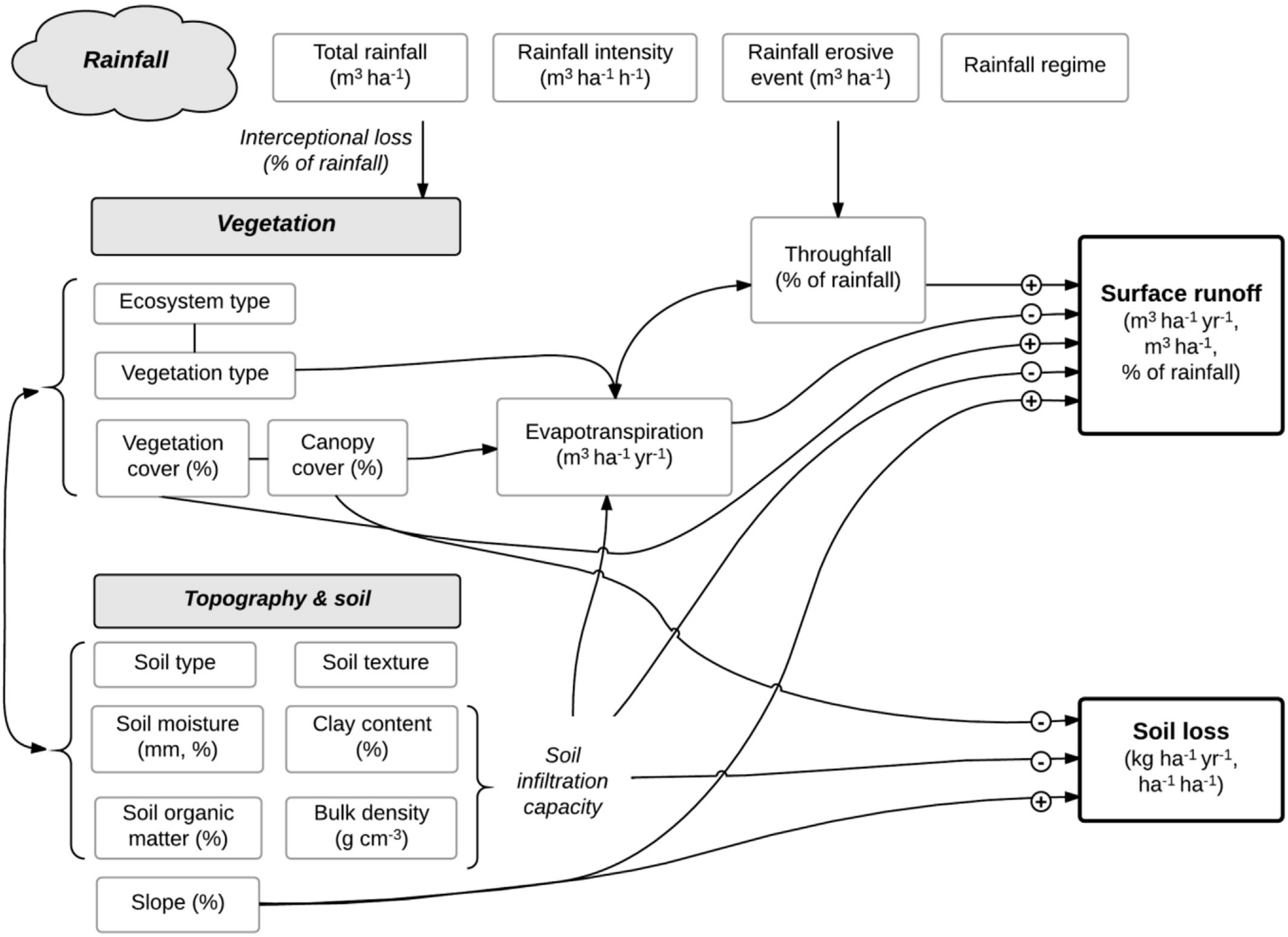

Fig. 3. Indicator interaction diagram for all the indicators approximating soil erosion and surface runoff. Their units are listed between parentheses.

scrublands co-occurs with high canopy cover, which results in lower runoff and soil loss than in grass-dominated ecosystems (Silburn, 2011).

Below the vegetation, the 'soil infiltration capacity' determines soil loss and surface runoff (Descheemaeker et al., 2006). This capacity is difficult to measure and mostly approximated by interrelated indicators such as 'soil moisture', 'clay content', 'soil organic matter' and 'soil bulk density' (Snyman, 1998; Vásquez-Méndez et al., 2010). The individual relationships between these factors and soil loss and runoff vary per vegetation and soil type. In general, high soil organic matter combined with low soil bulk density increase infiltration capacity (Snyman, 1998). The soil-related indicators have limited predictive value in isolation, but, once combined, they usefully approximate the soil infiltration capacity. Finally, slope is a major factor that increases soil loss and surface runoff (Vásquez-Méndez et al., 2010; Fu et al., 2011). The role of slopes (measured in degrees or percentage) on both processes is more important in sparse vegetation than in dense vegetation (Descheemaeker et al., 2006).

5. Results

Our analysis provided quantitative mean values for soil loss and surface runoff, for all eleven management regimes (Table 3). *High intensity grazed rangelands* were by far the most common management regime: 58% of the data entries on soil erosion and 36% on surface runoff related to this management regime. Other prevalent management regimes among soil erosion and surface runoff studies were *ungrazed natural rangelands* (10% each) and *low intensity grazed rangelands* (8% and 11% respectively). Both analyses yielded little data for *overgrazed rangelands* and data for the two *abandoned rangeland* regimes was especially limited for soil erosion. No surface runoff data was available for *abandoned degraded rangelands*.

5.1. Soil erosion and surface runoff per management regime

Table 3 shows some clear trends for both soil erosion and surface runoff. Compared to *ungrazed natural rangelands*, both soil loss and surface runoff increase notably with increasing grazing intensity.

Mean annual soil loss in *high intensity grazed rangeland, silvo-pasture* and *man-made pasture* differed substantially. Annual soil loss in *low intensity grazed rangeland* was notably lower than in more intensive livestock management regimes. The mean soil loss in *overgrazed rangeland* was extremely high, but this should be treated with caution because only two data entries were available. The high annual soil loss found for *silvo-pasture* results from including studies on intercropped trees with sown pastures rather than with natural rangelands. The limited data for such 'natural' *silvo-pastures* suggests annual soil loss values around or below that of *ungrazed natural rangelands* (McIvor et al., 1995). Interestingly,

Table 3
Mean values ($\overline{X}$) for soil loss and surface runoff per management regime. The standard error (SE) is given after each mean, followed by the number of data entries (n).

Management regime		Annual soil loss (kg ha^{-1} yr^{-1})		Soil loss (kg ha^{-1})		Annual surface runoff (m^3 ha^{-1} yr^{-1})		Surface runoff (m^3 ha^{-1})		Surface runoff coefficient (%)	
		$\overline{X}$	SE (n)	$\overline{X}$	SE(n)	$\overline{X}$	SE(n)	$\overline{X}$	SE(n)	$\overline{X}$	SE(n)
I.	Natural ungrazed rangeland	717	388 (17)	244	88(11)	98	42(13)	73	24(11)	40	5(6)
	Conservation rangeland	*no data*		*no data*		508	0 (1)	*no data*		0 (1)	(1)
II.	Low intensity grazed	1370	648(9)	1385	372 (10)	170	43(8)	314	210(4)	29	7(14)
	Restoration rangeland	126	28(16)	*no data*		66	15(16)	*no data*		*no data*	
III.	High intensity grazed	4048	1517(22)	500	85(134)	505	113(19)	2227	705(12)	21	2(70)
	Overgrazed rangeland	9915	3105(2)	*no data*		810	264(3)	*no data*		22	9(3)
	Silvo-pasture	3348	1029(14)	*no data*		894	209(12)	236	0(1)	5	1(2)
IV.	Man-made pasture	4249	1529(7)	*no data*		919	267(7)	164	0(1)	*no data*	
	Exotic tree plantation	89	211(14)	*no data*		254	56(16)	761	341(2)	*no data*	
V.	Abandoned rangeland	2705	1275(2)	100	0(1)	143	84(2)	478	133(21)	16	3(4)
	Abandoned degraded rangeland	*no data*		90	8 (7)	*no data*		*no data*		*no data*	

soil loss in both *exotic tree plantations* and *restoration rangelands* is considerably lower than in most other regimes. Both regimes aim to prevent or restore soil erosion. The low soil loss in *restoration rangelands* can, however, mostly be attributed to the predominantly flat surface measurements. Surface runoff values follow a similar pattern to that of soil loss and observations in *silvo-pastures* and *restoration rangeland* apply for surface runoff values as well.

Little data was available for soil loss and surface runoff per unit of area (Table 3). Higher means for soil loss and surface runoff in *high intensity grazed* as compared to *low intensity grazed rangelands* seem of limited value since these measurements were based on different lengths of time. The mean runoff coefficient was slightly higher in *natural ungrazed rangelands* compared to other the regimes but did not differ much between more intensive management regimes.

5.2. Underlying indicators for soil erosion and surface runoff

We compiled mean values of underlying indicators for soil loss and surface runoff (Tables C1 and C2 in Appendix C) and conducted a further correlation analysis between a few indicators in relation to soil loss and surface runoff.

Poor data availability compromised observing trends in rainfall-related indicators linked to soil loss. Soil loss in *high intensity grazed rangelands* was generally related to high rainfall intensity (Table C1 in Appendix C), but this could not be reliably compared to other management regimes. Vegetation cover did not differ among management regimes, regardless of differences in soil loss between them. Vegetation cover was even among the highest in *high intensity grazed rangeland*. However, canopy cover was notably higher in regimes with low soil loss values (70–80% vs. 5–40%, Table C1). Additional correlation analysis among all data entries yielded a negative but not statistically significant correlation between canopy cover and soil loss.

Soil bulk density was 1.2 g cm^{-3} in *high intensity grazed* and 1.3 g cm^{-3} in *low intensity grazed rangelands*, while clay contents of the same management regimes were around 33% vs. 18%. Soil organic matter contents were generally lower in *high intensity grazed* as compared to *low intensity grazed rangelands*. An additional correlation analysis of all data showed a considerable negative correlation between soil organic matter and soil loss (Spearman's $\rho = -0.757$, sig (2-tailed) < 0.01). Finally, the slopes of various management regimes differed substantially. Soil loss was measured at *ungrazed*, *restoration* and *high intensity grazed rangelands*, of which the slopes were steeper compared to other management regimes. These steep slopes can explain the high soil loss values because we collected many data entries for these *high intensity grazed rangelands*. However, the slopes were gentler for other regimes with both high and low soil loss, which suggests that this bias is limited. Additional correlation analysis showed a positive correlation between slope and soil loss (Spearman's $\rho = 0.386$, sig (2-tailed) < 0.01).

Similar to soil loss, we could not observe trends in rainfall-related indicators for surface runoff, again due to limited data availability (Table C2 in Appendix C). This suggests that most studies report runoff either without referring to actual rainfall or by linking it immediately to the aridity zone or annual rainfall statistics. Vegetation cover did also not differ conclusively among management regimes, regardless of differences in their runoff. Canopy cover was notably higher in regimes with low surface runoff values (70–90% vs. 5–40%). Additional correlation analysis among all data entries yielded a negative but not statistically significant correlation between canopy cover and surface runoff. Indicators for soil variables also showed similar trends as compared to the soil loss analysis. Limited data on soil bulk density showed again fractionally lower values in *high intensity grazed* as compared to *low intensity grazed rangeland* (1.2 vs. 1.3 g cm^{-3}). Clay contents of these management regimes were around 31% vs. 18%. Soil moisture contents were fractionally higher in *natural rangelands* as compared to other management regimes. Slopes of all management regimes were generally steeper compared to where soil loss had been measured. Notable slope differences occurred between *high intensity grazed rangeland* (12%) and *low intensity grazed rangeland* (5%), but this difference alone is unlikely to alter surface runoff. For instance, runoff in *abandoned rangelands* was measured at a 49% slope on average, but this runoff was only fractionally higher than that of *ungrazed rangelands*. Additional correlation analysis showed a positive correlation between slope and surface runoff (Spearman's $\rho = 0.328$, sig (2-tailed) < 0.01). Insufficient data on evapotranspiration and throughfall prevented comparisons between the different management regimes.

Because of notable differences in slope between the management regimes for both soil loss and surface runoff, we assessed all data for gentle slopes (less than 10%), which was the most common slope category. Although this reduced the number of data entries that could be analysed, it controlled for any exaggerated slope effects while still showing interesting trends. Table 4 shows that trends for annual soil loss and surface runoff follow largely the same pattern as the results for all slope categories showed above in Table 3.

5.3. Soil erosion prevention and water flow regulation as rangeland ecosystem services

The actual ecosystem services related to both soil loss and runoff (i.e. soil erosion prevention and water flow regulation) can be

Table 4
Mean values ($\overline{X}$) of soil loss and surface runoff for each management regime, but limited to slopes between 0 and 10%. The standard error (SE) is given after each mean, followed by the number of data entries (n). Abandoned degraded rangelands were excluded due to limited data.

Management regime		Annual soil loss (kg ha^{-1} yr^{-1})		Annual surface runoff (m^3 ha^{-1} yr^{-1})	
		$\overline{X}$	SE (n)	$\overline{X}$	SE (n)
I.	Natural ungrazed rangeland	950	550(2)	*no data*	
	Conservation rangeland	*no data*		*no data*	
II.	Low intensity grazed rangeland	1370	648(9)	171	43(8)
	Restoration rangeland	*no data*		*no data*	
III.	High intensity grazed rangeland	2338	719(18)	563	118(17)
	Overgrazed rangeland	9915	3105(2)	587	246(2)
	Silvo-pasture	3348	1029(14)	894	209(12)
IV.	Man-made pasture	4249	1529(7)	919	267(7)
	Exotic tree plantation	899	210(14)	255	56(16)
V.	Abandoned rangeland	2705	1275(2)	143	84(2)

determined by comparing the different indicators' values across various ecosystems with different naturalness and degradation levels (Bartley et al., 2006; Fu et al., 2011). For instance, soil loss of different land-use types is often compared to that of bare soil to determine soil erosion prevention capacity. In our study we consider both soil loss and surface runoff relative to the natural reference (i.e. *natural ungrazed rangeland*) as the potential ecosystem service (sensu Vásquez-Méndez et al., 2010). Based on soil loss and surface runoff of different management regimes relative to *natural ungrazed rangeland* and each other (Table 3), we can formulate the potential provision of the ecosystem services 'soil erosion prevention' and 'water flow regulation'. Mean values for annual soil loss in the *man-made pastures*, *high intensity grazed* and *low intensity grazed rangelands* regimes were, respectively six, five and two times higher than *natural ungrazed rangelands*. Surface runoff was, respectively nine, four and two times higher. Moreover, soil loss and surface runoff was reduced in *abandoned* and *restoration rangelands*. Altogether, these results suggest that potential soil erosion prevention and water flow regulation can be provided by reducing grazing intensity and active rangeland restoration. However, these findings should be treated with caution because the differences between management regimes were not tested statistically and study bias may occur. The large standard errors of the means and differences in number of data entries per management regime should be acknowledged (Table 3).

Fig. 4 shows the results of a further analysis performed with only the studies that compared soil loss and surface runoff between one or more management regimes. Changes in soil loss could be derived from eight studies, while only three studies compared surface runoff between management regimes. The results in Fig. 4 offer preliminary insights in soil erosion prevention and water regulation involved in changing from one regime to another. Soil loss increases notably from, respectively, *natural ungrazed* and *low intensity* to *high intensity* grazed rangelands (Fig. 4 A). *Exotic tree plantations* reduce soil loss of *man-made pasture* and *silvo-pasture*. Abandonment of *low intensity*, *high intensity* and *overgrazed rangelands* involves stark reductions in soil loss. Similar trends could be observed for surface runoff (Fig. 4 B).

6. Discussion and conclusion

6.1. A comprehensive typology of management regimes for rangelands

Definitions of management regimes for semi-arid to sub-humid rangelands were based on generic management indicators that reflect livestock grazing intensity, rangeland restoration or conservation. We selected indicators for differences in land-use intensity and key management activities. The indicators are applicable in all rangelands and clearly separate different management regimes. All indicators were qualitative or binary entities, for example stocking rate (high, low), soil treatment (yes/no) and planted, natural vs. exotic vegetation. These coarse indicators yielded comprehensive management regimes reflecting clear and relevant rangeland management decisions.

Developing the typology required several simplifications and this means that certain management indicators could not be considered. Assuming that rangeland management will have the same effect on all different vegetation types of semi-arid and sub-humid rangelands would be a generalization. The characteristics of the regimes reflect the strong consensus on the general effects of no, low and high livestock grazing on vegetation cover (Bellamy and Brown, 1994; Fynn and O'Connor, 2000; Stringham et al., 2003). However, the typology may overlook differences of the impacts on the variety of rangeland ecosystems (Dunkerley, 2000; MA, 2005). Similarly, responses of species composition and biodiversity to management could not be accounted for. In rangeland ecology, rangeland productivity and water use efficiency are frequently used for indicating livestock grazing impacts (Perevolotsky and Seligman, 1998; Allsopp et al., 2007). We incorporated them into a more generic indicator called 'rangeland condition', which has negative correlation with livestock grazing intensity (Snyman, 1998; Jouven et al., 2010). We did not incorporate highly specific livestock management indicators, such as mowing frequency, fire management frequency, irrigation and additional feeding (Perevolotsky and Seligman, 1998; Fynn and O'Connor, 2000). These would indicate subtle variations of already intensively used rangelands (i.e. *high intensity use rangelands* and *man-made pastures*), but the indicators in our study are sufficient to separate these regimes from less intensive ones. We also did not incorporate the above discussed management indicators because they have rarely been used to quantify soil erosion and surface runoff.

The typology presented in this paper is an alternative to the many vaguely defined and even subjective categories in the rangeland literature (e.g. Dormaar et al., 1994; Mwendera and Saleem, 1997). In addition, many studies reported vegetation cover to be affected by several grazing intensities without specification (e.g. Allsopp et al., 2007). Because livestock grazing is the chief management pressure on rangelands, we only considered well-defined grazing intensities (i.e. defined relative to the rangeland's carrying capacity and natural productivity (Fynn and O'Connor, 2000; Stringham et al., 2003)). Hence, we ignored highly variable grazing intensities or regime transitions. We note that the grazing duration and its location play an important role as most rangelands are highly adaptable to different grazing intensities (Perevolotsky and Seligman, 1998). We also came across different approaches to restore (overgrazed) rangelands, which could be characterized by the timing and duration of discontinued grazing, and the

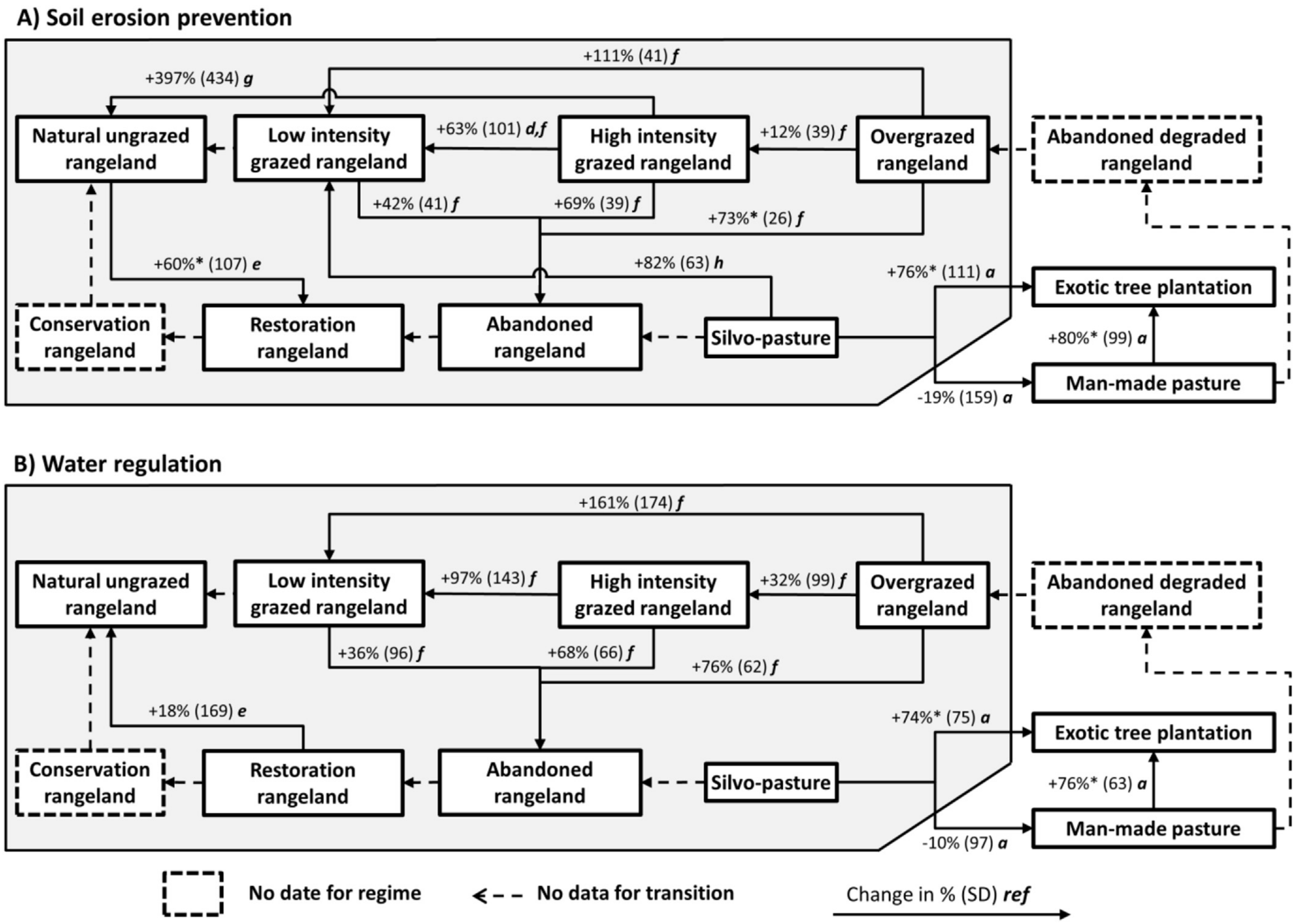

Fig. 4. Erosion prevention (A) and water flow regulation (B) involved in transitions between management regimes of semi-arid and sub-humid rangelands. Solid arrows indicate change in soil erosion prevention (A) and water regulation (B) between two regimes; positive numbers indicate erosion prevention and runoff reduction. Asterisks (*) indicate significant differences ($P < 0.05$). Some unquantified transitions (dashed lines) were omitted to improve the figure's readability. Note: Results are based on a subset of the data as presented in Table 3. Data sources are indicated with letters: a – Narain et al. (1997), b – Lechmere-Oertel (2003), c – Snyman and Van Rensburg (1986), d – Snyman (1999), e – Andreu et al. (1998), f – Mwendera and Saleem (1997), g – Helldén (1987), h – de Aguiar et al. (2010).

restoration degree (e.g. Launchbaugh, 1955; Dormaar and Willms, 1998; Muñoz-Robles et al., 2011). This resulted in the management regimes *conservation* (any grazing disabled, long-term conservation), *restoration* (active restoration, including replanting and removing unwanted vegetation), *silvo-pasture* (planting or leaving trees to reduce erosion), *exotic tree plantation* (same purpose) and *abandoned rangeland* (disabling grazing to let vegetation recover). Several of these management regimes could shift into other regimes. Our typology was expedient in defining these transitions unambiguously.

6.2. Soil erosion and surface runoff per management regime – a test of the typology

The results of our analysis of soil erosion and surface runoff per management regime can be seen as a test of how suitable our typology is for quantifying management effects. A challenge for such typologies is to determine whether generalised categories provide results that are precise and reliable enough to adequately mimic regional management effects on soil erosion and surface runoff (Stringham et al., 2003). The comprehensive set of management indicators enabled us to link quantified information to a specific management regime via simple cross-tabulation (Table 1). Most data were found for *high intensity grazed rangelands*, followed by *ungrazed natural* and *low intensity grazed rangelands*, *silvo-pasture* and *exotic tree plantations*. Although *conservation, abandoned degraded* and, to a lesser extent, *overgrazed rangelands* were frequently mentioned in the literature, very few quantitative assessments of either soil erosion or surface runoff could be found. The only information for *conservation rangelands* referred to underlying soil-related indicators (Launchbaugh, 1955; Dormaar et al., 1994). Limited information was found for *abandoned degraded rangelands*, as most studies focused on management regimes in transition to this regime (i.e. *high intensity grazed* or *overgrazed rangelands*). We note that *abandoned degraded rangelands* are frequently found in semi-arid rangelands, where woody encroachment and desertification are major problems (e.g. Muñoz-Robles et al., 2011; Puttick et al., 2011; Manjoro et al., 2012). Although many studies claim to study *overgrazed rangelands*, we found them to mostly focus on *high intensity grazed rangelands instead*. This can be attributed to the frequently used subjective definitions of overgrazing without truly assessing the ecological consequences and new equilibria that can evolve even after heavy grazing.

We calculated mean annual soil loss and surface runoff per management regime, but could not statistically test differences between regimes due to large differences in number of studies and data entries per regime, which would have resulted in study bias. The results, nevertheless, indicate clear trends in soil loss and surface runoff amounts in *natural ungrazed*, *low intensity grazed* and especially *high intensity grazed rangelands*. We found data from multiple studies in different regions for all management regimes except for *conservation, restoration* and *overgrazed rangelands*. Studies on *silvo-pastures* included both natural and man-made *silvo-pastures* and more research is needed to retrieve differences between those two entirely different management systems (McIvor et al., 1995). Furthermore, although few other correlations could be established, we found that slope is positively correlated to both soil loss and surface runoff, and that soil organic matter was strongly negatively correlated with soil loss. Soil organic matter and slope are, therefore, useful indicators for quantifying soil erosion and surface runoff. Our preliminary analysis of soil erosion prevention and water flow regulation involved in transitions between management regimes should be considered a first step towards establishing robust relations between rangeland management and these ecosystem services. Future research should focus on compiling information for meta-analyses based on multiple sources per management regime transition.

Other reviews on erosion and surface runoff mainly focused on the impacts of broad land-use types, such as cropland, livestock grazing in general (mainly high) and different forms of agriculture. Our review did not consider cropland, because cropland management in semi-arid and dry sub-humid regions tends to be localised, variable and mostly occurring in already converted systems (Kosmas et al., 1997). Considering cropland management would have resulted in even more management regimes in the *converted rangelands* regimes and, potentially, fewer data per management regime to analyse. Transitions between grazed rangelands and rangelands converted to croplands are understudied (Vásquez-Méndez et al., 2010). Moreover, livestock grazing has been identified as the chief pressure of land degradation in semi-arid and dry sub-humid regions (UN, 2011). We were, however, able to retrieve important indicators for both soil erosion and surface runoff from cropland studies (Kosmas et al., 1997; Fu et al., 2011). Although not all of these management practices apply to our typology, especially the ecological indicators could be used to develop our indicator overview and resulting indicator interaction diagram for soil erosion and surface runoff.

Our indicator overview and the interaction diagram offer an extensive overview of the key indicators for soil erosion and surface runoff. We did not include the standardized factors of the Universal Soil Loss Equation (USLE, c.f. Fu et al., 2011), because we only used measured values. We acknowledge that runoff is also only a small part of the hydrological cycle (Cerda et al., 1998; Bartley et al., 2006). Runoff is also relatively simply to measure unambiguously and the merits of reducing runoff are widely acknowledged in the literature, such as improving productivity and downstream water quality (Narain et al., 1997; Fu et al., 2009).

We did not assess the effects of management regimes on other ecosystem services than soil erosion prevention and water flow regulation. Despite the regimes' positive effects on soil erosion and surface runoff prevention, exotic tree plantations likely have negative effects on rangelands' biodiversity and ecosystem services. Rangelands are biodiverse and provide many different ecosystem services, such as medicinal plants, raw materials, tourism, and carbon sequestration (Mortimore, 2009). We note, however, that none of the erosion or surface runoff studies assessed the consequences to or trade-offs with providing other ecosystem services. Even services that are directly related to livestock grazing (e.g. fodder, milk, meat and wool), affected by soil erosion and/or runoff (e.g. water purification and soil fertility) or rangeland restoration (e.g. tourism, habitat for large grazers and carbon storage), have rarely been assessed in the literature. An apparent conclusion is that soil erosion and surface runoff have thus far only been assessed in high detail by 'traditional' disciplinary soil scientists and hydrologists, whereas the merits of preventing these processes have been largely neglected.

6.3. Conclusion

We assessed the consequences of management decisions in semi-arid to sub-humid rangelands by studying the effects of management regimes on soil erosion and surface runoff. Our results show that both soil loss and surface runoff are high in management regimes with high livestock grazing intensity. Soil loss and surface runoff reduced in management regimes that aim to reverse land degradation of intensive grazing (i.e. *abandoned* and *restoration rangelands*). Our further analysis, a preliminary assessment of transitions between management regimes confirmed that increasing livestock grazing intensity indeed increases soil erosion and surface runoff. Moreover, soil loss and surface runoff are reduced considerably when *man-made pastures* are converted to *exotic tree plantations* and if *intensively grazed rangelands* are abandoned. Our findings suggest that management can reverse land degradation involved in all management regimes apart from *degraded abandoned rangelands*. Moreover, our research underlines the risks involved in intensifying livestock grazing in semi-arid and sub-humid rangelands.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jaridenv.2015.05.015.

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