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A global meta-analysis on the monetary valuation of dryland ecosystem services: The role of socio-economic, environmental and methodological indicators

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ABSTRACT

Monetary valuation of dryland ecosystem services may help to increase the salience of drylands in decision making. Yet, there is no comprehensive assessment of the indicators that determine the estimated monetary values for dryland ecosystem services (hereafter: dryland value). Having compiled a database consisting of 559 observations from 66 valuation studies in drylands worldwide, this study analyzes the relative importance of local socio-economic, environmental and methodological indicators in explaining the monetary value estimates for nine dryland ecosystem services by means of a multiple regression analysis. By explicitly quantifying the effect sizes of the indicators of dryland value, we shed new light on the driving forces behind monetary valuation of dryland ecosystem services. Our results show that local socio-economic and environmental conditions are marginal in explaining dryland value, indicating that local dryland conditions are not sufficiently captured with current valuation approaches. Simultaneously, we find that methodological factors, including valuation method and study extent, heavily influence dryland value, suggesting that monetary valuation outcomes are largely determined by the selected methodology. This emphasizes the need to improve monetary valuation methods so that they better capture local dryland conditions in order to be able to serve as a meaningful tool for decision making.

1. Introduction

Covering about one third of the global land surface, drylands are a critical biome for about one third of the global human population (Fig. 1; Bastin et al., 2017; MA, 2005; Reynolds et al., 2007), who depend on an extensive set of ecosystem services for their wellbeing and livelihood (Boafo et al., 2016; Favretto et al., 2016; MA, 2005). However, because drylands – that are defined by a 0.05–0.65 degree of aridity (Leemans and Kleidon, 2002; UNCCD, 1994) – are typically located in the least developed regions of the world, they have thus far received little attention in public opinion and environmental policy and decision making (Reynolds et al., 2007; Thomas et al., 2012). In recent years, it has been proposed that the estimation of monetary values for ecosystem services may be a tool to increase the salience of such services in decision making processes (Daily et al., 2009; Fisher et al., 2008). With regard to drylands, such information may, for example, be useful to recently launched initiatives, such as the Land Degradation Neutrality concept adopted by the UNCCD (Orr et al., 2017), the Economics of Land Degradation initiative (ELD, 2015) and the IPBES assessment on land degradation and restoration (IPBES, 2017; Opgenoorth and Faith, 2013). Monetary valuation may, for instance, help to better account for the costs of land degradation and the benefits of sustainable land management in decision making (Quillérou and Thomas, 2012; Turner et al., 2016).

Yet, although monetary valuation of ecosystem services aims to estimate the societal benefits of ecosystem services that accrue to their

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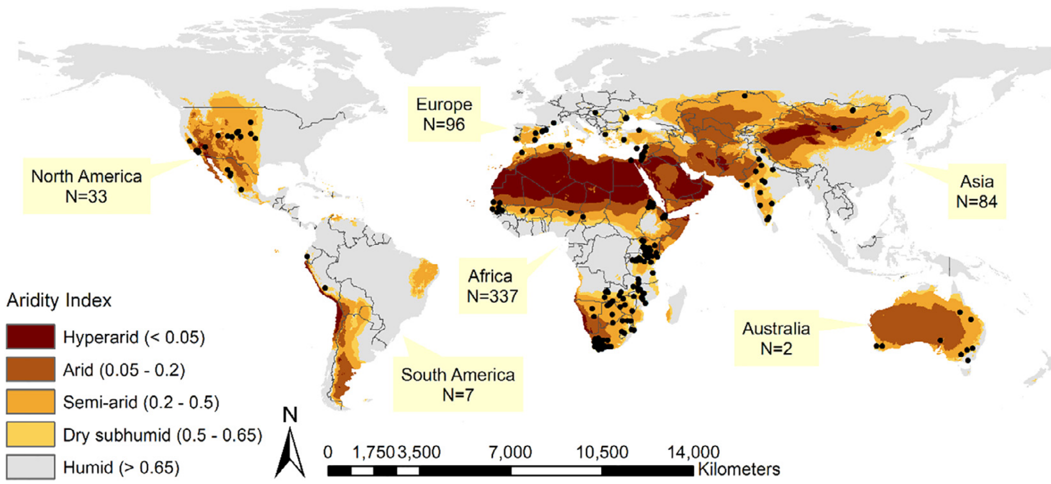

Fig. 1. Global map of aridity, indicating arid, semi-arid and dry subhumid land zones (derived from FAO, 2009), which shows the geographical locations of the dryland study sites ($N = 204$) where observations are located that have been summarized in the database of this study ($N = 559$). The number of observations per continent is indicated. The map has a spatial resolution of 10 arc minutes and temporal coverage of 1961–1990.

beneficiaries (Bateman et al., 2011; Daily et al., 2009; Heal, 2000) and is the most widely used method for ecosystem services valuation up to date (de Groot et al., 2012; Liu et al., 2010), the approach is also widely criticized. Several studies, for instance, suggest that monetary valuation may have difficulty to capture ecosystem dynamics (Farley, 2012; Polasky and Segerson, 2009) and that the researchers' selection of the study scope and methodology have a large influence on the valuation outcome (Martín-López et al., 2014; Schild et al., 2017; Spangenberg and Settele, 2010). For instance, while meta-analyses in other biomes find varying support for the role of socio-economic conditions, they all find evidence of the importance of methodological factors (Brander et al., 2006; De Salvo and Signorello, 2015; Enjolras and Boisson, 2008; Ghermandi et al., 2010; Quintas-Soriano et al., 2016; Salem and Mercer, 2012; Woodward and Wui, 2001). If methodological factors are more important than local conditions with regard to ecosystem properties and socio-economic conditions of beneficiary populations, this suggests that monetary valuation of ecosystem services does not (yet) deliver on its promise.

Despite the critiques, the number of monetary valuation studies has been growing rapidly in the last decades (Liu et al., 2010). This also holds for drylands, although only a few studies explicitly mention that they focus on dryland valuation (Barrow and Mogaka, 2007; Birch et al., 2010; Hein, 2007; O'Farrell et al., 2011). The growing attention for monetary valuation increases the relevance of testing whether such valuation studies do actually capture socio-economic and environmental factors, as they are supposed to do. Drylands are a good case to test this, because their inhabitants are particularly vulnerable to environmental degradation and the associated loss of ecosystem services needed for subsistence (Cowie et al., 2011; Stafford Smith et al., 2009; Verstraete et al., 2009), which should ideally be reflected in the estimated value. As it is difficult to generalize from individual valuation studies alone, amongst others because of their limited geographical focus, the best way to analyze whether the critiques hold is by conducting a meta-analysis, which allows to assess general trends and patterns (Nelson and Kennedy, 2008). To our best knowledge, such a meta-analysis focused on the monetary valuation of dryland ecosystem services has not been carried out so far.

In order to address this research gap, we have identified and compiled valuation studies that estimated the monetary value of ecosystem services in drylands (hereafter: dryland value), resulting in a comprehensive database of dryland value observations. In order to analyze which indicators determine dryland value, we complemented the database with indicators for local socio-economic, environmental and methodological conditions. We hypothesized that local socio-economic conditions would be relevant, as the welfare of ecosystem service beneficiaries is predominant in determining their values, which may particularly apply for drylands due to the marginalized status of their inhabitants. We also hypothesized that local environmental conditions explain a substantial proportion of the variance in dryland value, because the supply of ecosystem services depends on underlying ecosystem functioning (de Groot et al., 2002), which may be particularly vulnerable to critical degradation thresholds in case of drylands (Verstraete et al., 2009). Lastly, we hypothesized that differentiation in estimate monetary values exists among dryland ecosystem services and dryland ecosystem types, as the dryland biome encompasses a wide range of ecosystems, each having their own distinctive processes and functions.

This meta-analysis contributes to literature on monetary valuation of ecosystem services in three different ways. First, our study is the first that comprehensively analyzes for drylands what indicators determine the estimated monetary values of ecosystem services. Second, while previous studies in other biomes focused mainly on socio-economic and methodological predictors of ecosystem service value estimates and often did not directly address environmental factors (Brander et al., 2006; Ghermandi et al., 2010; Johnston et al., 2005), we include an extensive set of (dryland relevant) environmental indicators in order to investigate to what extent they determine the monetary value estimates for dryland ecosystem services. Third, compared to previous studies in other biomes, this study is the first to explicitly quantify the relative importance (i.e. effect size) of various indicators in determining monetary value estimates for ecosystem services. In addition to these contributions to the literature, our empirical analysis of the drivers of monetary valuation of dryland ecosystem services may also have implications for the meaningfulness of their use in policy making, especially with regard to recent initiatives.

2. Methods

2.1. Compilation of the dryland value database

To compile a database with observations on dryland value, monetary valuation studies of ecosystem services that were located in drylands were collected using two different approaches: (1) valuation studies that were located in drylands were identified from the TEEB database (van der Ploeg and de Groot, 2010), and (2) valuation studies were collected from a literature search in grey and peer-reviewed literature. For valuation studies that were identified from the TEEB database, all original valuation studies were retrieved. As the number of dryland valuation studies identified from the TEEB database was limited, the literature search was carried out to retrieve more dryland valuation studies, by searching in online search engines, relevant reference lists and bibliographies, and non-English sources (last search dates from: 31/03/2015). The collected valuation studies from this literature search included (peer-reviewed) journal articles, book chapters, conference papers and reports for public institutions. Together, this resulted in 66 collected valuation studies (see Appendix Table A.1 for an overview of these studies).

Observations from the collected valuation studies were only included in the dryland value database when they met the following requirements: (1) the study site was located in a dryland, which is defined by a degree of aridity between 0.05–0.65 (i.e. including arid, semi-arid and dry subhumid climate zones; see Fig. 1; Leemans and Kleidon, 2002; UNCCD, 1994), (2) the estimated value for an ecosystem service represented a monetary value which could be converted into a standardized value, and (3) sufficient data characteristics were available to determine relevant indicators for this study. For all observations that were collected from the TEEB database, original valuation studies were inspected in order to improve the recorded information for these observations and supplement them with additional data for the indicators relevant for this study. The resulting dryland value database included 559 observations collected from 66 valuation studies.

For each observation of dryland value in the database, data was collected for indicators related to the valuation study, including ecosystem service, ecosystem type, valuation method, study areal extent (in hectare) and year of valuation (Table 1). For ecosystem service, ecosystem type and valuation method, the classification by TEEB was followed (de Groot et al., 2010; Pascual et al., 2010). For ecosystem service, a few categories of ecosystem services were aggregated to create ecosystem service groups that had a sufficient number of observations for robustness in the statistical analysis (Table 2). Bundling of some these ecosystem services into groups may have led to a larger variance in value (e.g. for biological regulation), but was not found to have major impact on the results, as other ecosystem services groups had larger variances (e.g. food provision). Furthermore, due to the dryland context of our study, several subservices (i.e. natural dyes, oils and salts) placed within the raw materials group by the TEEB classification, fitted better in the biochemicals provision group and were therefore reclassified accordingly (Table 2). Together, this resulted in the following dryland ecosystem service groups: food, fresh water, raw materials and biochemicals provision, climate, water, soil and biological regulation, and cultural services (Table 2).

Furthermore, the ecosystem type and valuation method were explicitly recorded for each observation in the dryland value database. For ecosystem types, the TEEB classification was adapted to specifically fit ecosystem types that are commonly identified in drylands (MA, 2005; Maestre et al., 2012; Scoones, 1991). These included semi-deserts, grasslands, woodlands, dry forests, (semi-)arid wetlands (hereafter: arid wetlands) and cultivated land (see Appendix Table A.2 for a detailed description of each ecosystem type). Although the occurrence of arid wetlands may seem counterintuitive in drylands, they have been widely documented in drylands, being either of a temporary or permanent nature (Scoones, 1991; Williams, 1999). For valuation method, methods were categorized into market pricing, production function, cost-based, travel cost, contingent valuation, benefit transfer and other methods (see Appendix Table A.3 for a detailed description of each of these methods). A comprehensive description of each of these monetary valuation approaches can be found in Bateman et al. (2011), Farber et al. (2006) and Freeman III (2003).

Monetary estimates for dryland value were standardized to 2007 International Dollar per hectare per year (hereafter: Int$/ha/yr) in order to have standardized values with a consistent currency for values that originated from different countries and were estimated for different years. We arrived at 2007 International Dollar per hectare per year values through the following steps. First, when studies reported monetary values estimated in foreign currencies, they were recalculated to their local currency value using the official exchange rate for the original year of valuation. Second, in order to correct for differences in purchasing power between countries, the local currency values were converted to International Dollars using the Purchasing Power Parity (PPP) conversion factor. Third, the International Dollar values were standardized to the year 2007 using the GDP deflator in order to correct for price inflation between years. The data on the official exchange rate, PPP conversion factor and GDP deflator were all obtained from World Bank Development Indicator databases (World Bank, 2010).

2.2. Collection of local socio-economic and environmental conditions

In order to analyze the role of local socio-economic and environmental conditions in determining dryland value, data on a variety of indicators that were relevant within the context of dryland ecosystem services valuation was collected for each dryland value observation in the database (Table 1; Sommer et al., 2011; Verstraete et al., 2011). The 559 observations for dryland value in the database came from 204 different study sites that were spread across drylands globally (Fig. 1).

Table 1
Variables that were collected in the dryland value database (N = 559) and included in the regression analysis.

Variable	Indicator	Unit	Data source
Ecosystem service		classes	Original valuation studies
Ecosystem type		classes	Original valuation studies
Valuation method		classes	Original valuation studies
Study areal extent	Study extent	ha	Original valuation studies
Land Use System	Land use	classes	Land Use Systems of the World (LADA, 2008)
Human Appropriation of Net Primary Productivity (HANPP)	Land use intensity	gC/m^2/ha	Haberl et al. (2007)
Regional GDP per capita	Population welfare	Int$ 2007	Kummu et al. (2018) and Gennaioli et al. (2013)
Regional population density	Population pressure	people/km^2	Gridded Population of the World (GWP-V4; CIESIN, 2016)
Degree of aridity	Water availability		Local Climate estimator (FAO, 2010)
Leaf Area Index (LAI)	Vegetation cover	m^2 leaf/m^2 ground	MOD15A2H (Myneni et al., 2015)
Gross Primary Productivity (GPP)	Vegetation productivity	gC/m^2/ha	MOD17A3 (Zhao and Running, 2010)
Soil pH	Soil acidity		Harmonized World Soil Database (HWSD; FAO/IIASA/ISRIC/ISSCAS/JRC, 2012)
Soil organic C content	Soil fertility	% weight	Harmonized World Soil Database (HWSD; FAO/IIASA/ISRIC/ISSCAS/JRC, 2012)
Soil Available Water Capacity (AWC)	Soil moisture content	mm/m	Harmonized World Soil Database (HWSD; FAO/IIASA/ISRIC/ISSCAS/JRC, 2012)
Soil sodicity	Soil crusting	%	Harmonized World Soil Database (HWSD; FAO/IIASA/ISRIC/ISSCAS/JRC, 2012)

Table 2
Dryland ecosystem services in the dryland value database (N = 559).

Ecosystem service type	Ecosystem service group[a]	Description
Provisioning	Food provision	Fish (N = 16), meat (N = 22), vegetables (N = 29) and forest food products (N = 32)
	Fresh water provision	Water for drinking (N = 11), irrigation (N = 3), industrial (N = 2) and general use (N = 7)
	Raw materials provision	Timber (N = 28), fuelwood and charcoal (N = 36), fibers (N = 23), fodder (N = 54) and other bulk materials (N = 6)
	Biochemicals provision	Genetic (N = 12), medicinal (N = 10) and ornamental resources (N = 20), food spices, supplements and other non-timber forest products (N = 20)
Regulating	Climate regulation	Carbon sequestration (N = 40)
	Water regulation	Water flow regulation (N = 34), water purification (N = 9) and flood attenuation (N = 2)
	Soil regulation	Soil erosion prevention (N = 18) and maintenance of soil fertility (N = 4)
	Biological regulation	Biological control (N = 3), pollination (N = 5), nursery (N = 3) and maintenance of biological and genetic diversity (N = 37)
Cultural	Cultural services	Recreation (N = 28), (eco)tourism (N = 28), hunting (N = 11), aesthetic (N = 2) and inspirational services (N = 4)

[a] Following the ecosystem services classification by TEEB (de Groot et al., 2010), which was finetuned to fit ecosystem services that were specifically recorded for drylands in the database.

Although ideally average values for the entire study areas would have been collected, we were constrained to use the spatial midpoint of the study sites, as the specific geographical configurations of the study areas were not known. This way, data collection for the indicators was consistent across all observations. The data distribution of the collected indicators can be found in Appendix Fig. A.1.

For local socio-economic conditions, indicators for land use, land use intensity, population welfare and population pressure were recorded. For land use, the Land Use System by LADA (2008) was used, which incorporates both the major land use as well as the type of land management. Data was collected from a map with a spatial resolution of 5 arc minutes for the year 2010. Land use was categorized into seven classes: intensive agro-pastoralism, moderate agro-pastoralism, intensive pastoralism, extensive pastoralism, protected, unmanaged and urban (see Appendix Table A.4 for a description of each class). As an indicator for the intensity of land use, the Human Appropriation of Net Primary Productivity (HANPP, in gC/m^2/yr) was obtained from a map with a spatial resolution of 5 arc minutes for the year 2000 (Haberl et al., 2007).

As indicators for population welfare and pressure, we collected GDP per capita and population density, respectively. Because the spatial scale of the delivery of ecosystem services to beneficiaries is likely to be larger than just the local scale (Hein et al., 2006), we selected a regional spatial scale for data collection for these two variables. For regional GDP per capita, subnational data based on Purchasing Power Parity was extracted from a gridded global dataset for the year 2005 (and standardized to 2007 International Dollars) with a spatial resolution of 5 arc minutes (Gennaioli et al., 2013; Kummu et al., 2018). It should be noted that in this dataset national data has been used due to a lack of subnational data for some countries, mainly concerning West and Central African countries. Regional population density was obtained from the UN-adjusted population density map (‘Gridded Population of the World’ (GWP), version 4; in people/km^2) for the year 2005 (CIESIN, 2016). From this gridded map having a spatial resolution of 30 arc seconds, we calculated regional population density using the same subnational data distribution as used for regional GDP per capita data that was developed by Gennaioli et al. (2013).

For local environmental conditions, we collected indicators on water availability, vegetation cover, vegetation productivity and soil conditions, as these environmental properties play a key role in dryland functioning (D’Odorico and Bhattachan, 2012; Delgado-Baquerizo et al., 2013; Verstraete et al., 2011). As an indicator for water availability, the degree of aridity was calculated based on the ratio of annual average precipitation (P) over annual average potential evapotranspiration (PET; Leemans and Kleidon, 2002). This data was derived from the ‘*Local Climate Estimator*’ (New_LocClim, version 1.10), which provides local, spatially interpolated climate data (FAO, 2010). Because these aridity measurements did not take the effect of water transported from elsewhere into account, either from natural origin (e.g. upstream catchments) or from artificial origin (e.g. irrigation structures), all study sites were inspected whether they received water predominantly from allogenic or autogenic sources. When visualized in a plot of aridity against dryland value, no sign for any deviations in aridity measurements was observed (see Appendix Fig. A.2).

As an indicator for vegetation cover, annual average Leaf Area Index (LAI; in m^2 leaf/m^2 ground) was collected for the time period 2000–2016 with a spatial resolution of 2.5 arc minutes based on MOD15A2H products (Myneni et al., 2015). For vegetation productivity, annual average Gross Primary Productivity (GPP, in gC/m^2/yr) was extracted from a map for the time range 2000–2013 with a spatial resolution of 30 arc seconds based on MOD17A3 products (Zhao and Running, 2010). As indicators for soil conditions, different soil type variables were extracted from the Harmonized World Soil Database (having 30 arc seconds spatial resolution), including soil pH as an indicator for soil acidity, soil organic carbon (C) content (% weight) as an indicator for soil fertility, Available Water Capacity in the soil (AWC, mm/m) as an indicator for soil moisture content and soil sodicity (% exchangeable sodium) as an indicator for soil crusting (FAO/IIASA/ISRIC/ISSCAS/JRC, 2012).

2.3. Statistical analysis of dryland value

Multiple linear regression analysis was carried out to analyze the significance and relative importance of multiple variables in explaining variation in dryland value. The logarithm (^{10}log) of dryland value was used in order to meet the condition for the dependent variable to come from a normal distribution, as tested with the Shapiro–Wilk test (W = 0.99, p = 0.18). Consequently, only back-transformed log values are presented. In the regression analysis, ecosystem service, ecosystem type, valuation method and land use were treated as fixed factors. We also included three interaction terms for ecosystem service with each of the other categorical variables, being ecosystem service with ecosystem type, ecosystem service with valuation method and ecosystem service with land use. Furthermore, the following variables were treated as continuous variables in the regression analysis: study extent, HANPP, regional GDP per capita, regional population density, aridity, LAI, GPP, soil pH, soil organic C content, soil AWC and soil sodicity. Of these variables, ^{10}log transformations were used for study extent, regional GDP per capita, regional population density, soil organic C content and soil sodicity.

In the regression analysis, we used stepwise regression to select the best model fit for the data using Akaike Information Criterion (AIC) as a selection criterion. Regression coefficients (unstandardized) and their standard errors were calculated with the Ordinary Least Squares (OLS) method. Regression models were controlled for heteroscedasticity (i.e. through visual inspection of the standardized residuals plot) and multicollinearity (i.e. VIF < 10). Because of high collinearity between ecosystem service, ecosystem type, valuation method and land use, two different regression models were built: the first model included ecosystem service with ecosystem type and land use, while the second model included ecosystem service with valuation method (in addition to other explanatory variables).

For exploration of the two final regression models, we examined the relative importance of the regression coefficients by calculating their effect size, which is a measure that indicates the magnitude of the effect of a regression variable. Three measures of effect size were calculated to evaluate consistency among effect size metrics, including omega-squared (ω^2), partial eta-squared (η^2_p) and eta-squared (η^2). Furthermore, for both regression models we explored the variables that could significantly explain variation in dryland value. For continuous variables, we ran simple linear regressions against dryland value to analyze their relation with dryland value. For categorical variables, we carried out Tukey post hoc tests to analyze whether there were significant differences among the categories of these variables (tested at $p < 0.05$ level of significance).

2.4. Control analyses of the regression models

Several control analyses were carried out for the two final regression models. To control for independence of observations, that were either obtained from the same study or had the same first author, we visually evaluated whether particular observations had a strong impact on the results obtained (Nelson and Kennedy, 2008). Potential study bias was examined by plotting studies that had a high number of observations in our database ($N \geq 20$) in the standardized residuals plots of the two regression models. To examine potential author bias, observations with the same first author for whom a high number of observations was included in the database ($N \geq 20$) were plotted in the standardized residuals plots as well. In addition, as year of valuation included a few older observations in the dataset (i.e. ranging between 1976–2009) and had a slightly positive relation with dryland value (Appendix Fig. A.3), year of valuation was also added to our list of regression variables in the model selection, but it was not selected in any of the models due to its very low explanatory power ($R^2_{adjusted} = 0.006$). Furthermore, some study sites were only partly located within drylands (at least 50% of the study extent had to be located within a dryland to be included in the database in the first place). To ensure that these observations ($N = 75$) had no effect on the results, they were tested against observations that were located completely within drylands using a *t*-test. Lastly, we also used a *t*-test to test whether observations that came from grey literature studies did not differ significantly from observations that came from peer-reviewed studies in order to make sure that they had no effect on the results.

3. Results

3.1. General patterns in dryland value

In the dryland value database, mean estimated values for individual dryland ecosystem services ranged between 14 Int$/ha/yr for climate regulation and 218 Int$/ha/yr for fresh water provision and the mean values for all dryland ecosystem services summed up to 586 Int$/ha/yr together (based on back-transformed log values; Table 3). Water-related ecosystem services including fresh water provision and water regulation had relatively the highest mean values, i.e. 175 Int$/ha/yr and 218 Int$/ha/yr respectively. In contrast, food and raw materials provision and climate and soil regulation received lowest mean values (between 14–20 Int$/ha/yr). Food and biochemicals provision covered the largest value ranges, indicating large variation among values estimated for these ecosystem services.

3.2. Multiple regression analysis of dryland value

The multiple regression models explained between 30–40% of the variation in dryland value. The first model – which included ecosystem services with land use and ecosystem type – explained 40% of variation in dryland value ($R^2_{adjusted} = 0.40$, $F(103,455) = 4.56$, $p < 0.001$). In this model, significant regression variables included in order of relative importance: land use, the interaction between ecosystem service and ecosystem type, ecosystem service, the interaction between ecosystem service and land use, study extent, ecosystem type, aridity and soil pH (Fig. 2a). The second model – which included valuation method with ecosystem services – explained a smaller amount of variation in dryland value with fewer significant regression variables ($R^2_{adjusted} = 0.27$, $F(45,513) = 5.63$, $p < 0.001$). These regression variables included in order of relative importance: valuation method, study extent, the interaction between ecosystem service and valuation method, ecosystem service, HANPP and soil pH (Fig. 2b). Interestingly, regional population density, regional GDP per capita, LAI, GPP, soil organic C content, soil AWC and soil sodicity were not included in any of the regression models: they did not significantly contribute to explaining any additional variation in dryland value.

The two regression models showed no sign of heteroscedasticity or multicollinearity (*VIF* < 10). Additionally, control analyses did not show any sign of author or study bias for both models (see Appendix Fig. A.4 for the standardized residuals plots of the regression models). Furthermore, dryland values were virtually equal for observations from study sites that were located either partly ($N = 72$) or completely ($N = 487$) within a dryland ($t(557) = 0.01$, $p = 0.99$; Appendix Fig. A.5). Lastly, dryland values did not differ whether they came from peer-reviewed or grey literature studies ($t(557) = 0.21$, $p = 0.83$; Appendix Fig. A.6). Thus, no methodological concerns about the dataset were observed, indicating that the dataset constituted a solid basis for our analysis.

Table 3
Descriptive statistics of the estimated monetary values for dryland ecosystem services (expressed in 2007 Int$/ha/yr) that were summarized in the dryland value database ($N = 559$).

Ecosystem service	Mean[a]	S.D.[a]	Median[a]	Minimum[a]	Maximum[a]	N
Food provision	15.58	14.13	11.80	0.01	11,988.28	99
Fresh water provision	174.97	14.45	138.18	0.43	7209.50	23
Raw materials provision	19.17	8.71	18.07	0.10	5648.82	147
Biochemicals provision	37.39	29.51	33.72	0.01	78,323.12	62
Climate regulation	14.45	7.41	9.70	0.67	2200.58	40
Water regulation	217.92	6.76	251.48	4.70	10,472.85	45
Soil regulation	19.27	6.92	32.23	0.11	218.90	22
Biological regulation	41.49	21.88	35.91	0.03	9,901.07	48
Cultural services	45.61	9.55	33.29	0.96	6,102.69	73
Total	*585.84*	*119.32*	*564.38*	*0.01*	*78,323.12*	*559*

[a] Mean, standard deviation (S.D.), median, minimum and maximum values were back-transformed from ^{10}log values. N is the number of observations.

3.3. Relations of dryland value with specific determinants

Continuous variables that were significant in the regression models were – in order of the amount of variation explained – study extent, HANPP, aridity and soil pH. A large amount of scatter among the observations was observed, when depicted (Fig. 3). Study extent, which had a medium effect size (between 4 and 6% in the regression models), had a negative relation with dryland value ($F(1,557) = 45.34$, $p < 0.001$, $R^2_{adjusted} = 0.07$, $y = 724.44\ x^{-0.25}$; Fig. 3a). HANPP, which was significant in the second model with a small effect size (1.5%), had a positive relation with dryland value ($F(1,557) = 24.42$, $p < 0.001$, $R^2_{adjusted} = 0.04$, $y = 10^{1.18 + 0.002x}$; Fig. 3b). Aridity, which was significant in the first model with a small effect size of 0.4%, had a slightly positive relation with dryland value ($F(1,557) = 5.17$,

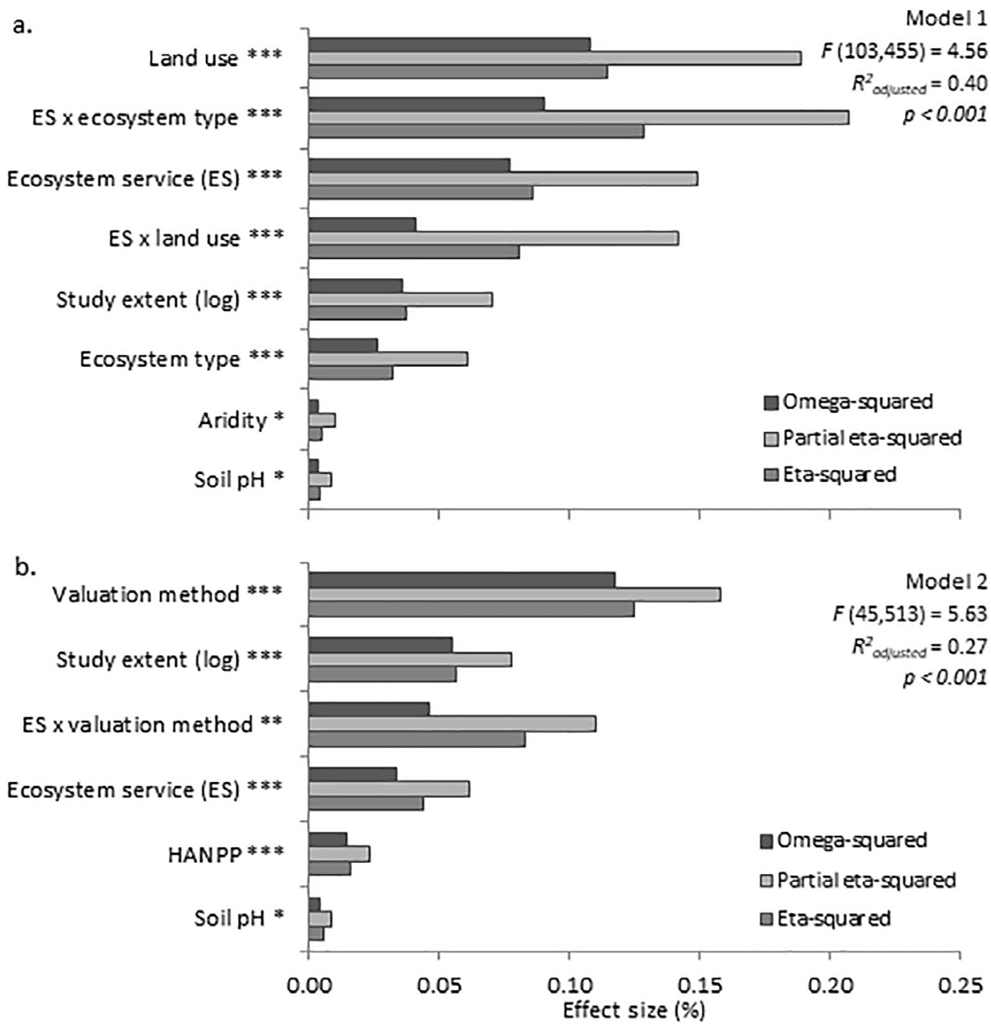

Fig. 2. Bar diagrams showing the relative importance of the significant regression variables (*y*-axes) sorted by effect size (*x*-axes, in%) for (a) regression model 1 on ecosystem service (ES) with land use and ecosystem type, and (b) regression model 2 on ecosystem service with valuation method. As measures for effect size, omega-squared (ω^2), partial eta-squared (η^2_p) and eta-squared (η^2) are depicted. For each individual regression variable, their level of significance in the regression models is indicated with ***, ** and * for 1%, 5% and 10% levels of significance, respectively. Regression statistics of both models are reported. Note that when ω^2 effect sizes of individual regression variables are summed up, the reported $R^2_{adjusted}$ values of the regression models are obtained.

p = 0.02, $R^2_{adjusted}$ = 0.007, $y = 10^{1.22\,+\,0.65x}$; Fig. 3c). For soil pH, which had a very small effect size in both models (0.3–0.5%), no significant relation with dryland value was found ($F(1,557)$ = 0.01, p = 0.91; Fig. 3d).

The significant categorical regression variables explained a substantial part of the variation in dryland value, being – in order of the amount of variation explained in the regression models – valuation method, land use, ecosystem service and ecosystem type in addition to the interactions with ecosystem service. These latter interaction effects have been analyzed in detail in Schild et al. (2017).

Valuation method was found to explain the largest part of variation, which accounted for an effect size of 12% in the second model. According to the post hoc test, market pricing and contingent valuation yielded significantly lower value estimates than production function and benefit transfer (Fig. 4a). The interaction between ecosystem service and valuation method was also significant and had a medium effect size (5%), which indicates that specific methods estimated monetary values for specific ecosystem services differently.

In addition, land use explained a large part of the variation (11%) in the first model. Intensive agro-pastoralism was found to be estimated significantly higher than any of the other land use classes (Fig. 4b). Land use also had a significant interaction effect with ecosystem service, which accounted for a smaller part of the explained variation (4%). This suggests that the monetary value for dryland ecosystem services is estimated differently when they originate from different land use classes.

Furthermore, ecosystem service individually had a medium effect size (8%) in the first model and a small effect size (3%) in the second model. Following the post hoc test, the monetary value for water regulation was significantly higher than any other service, except for fresh water provision (results for model 1 were reported, as this model explained the data best). In addition, the monetary value for fresh water provision was estimated higher than the monetary values for food provision and soil regulation (Fig. 4c).

Lastly, the interaction effect between ecosystem service and ecosystem type had a large effect size in the first model (9%). This interaction indicates that the monetary value of ecosystem services that are provided by different ecosystem types were estimated differently. Ecosystem type individually had a small effect size in the same model (3%). Among ecosystem types, cultivated land was found to be estimated significantly higher than grasslands, woodlands and wetlands. Also, semi-deserts had a higher value estimate than grasslands (Fig. 4d).

4. Discussion

In this study, we analyzed which factors determine the monetary values that have been estimated for dryland ecosystem services in valuation studies. Here, we discuss the main trends and patterns that emerged from this meta-analysis.

4.1. Water-related ecosystem services in drylands

We found that the monetary value for water-related ecosystem services, including fresh water provision and water regulation, was high

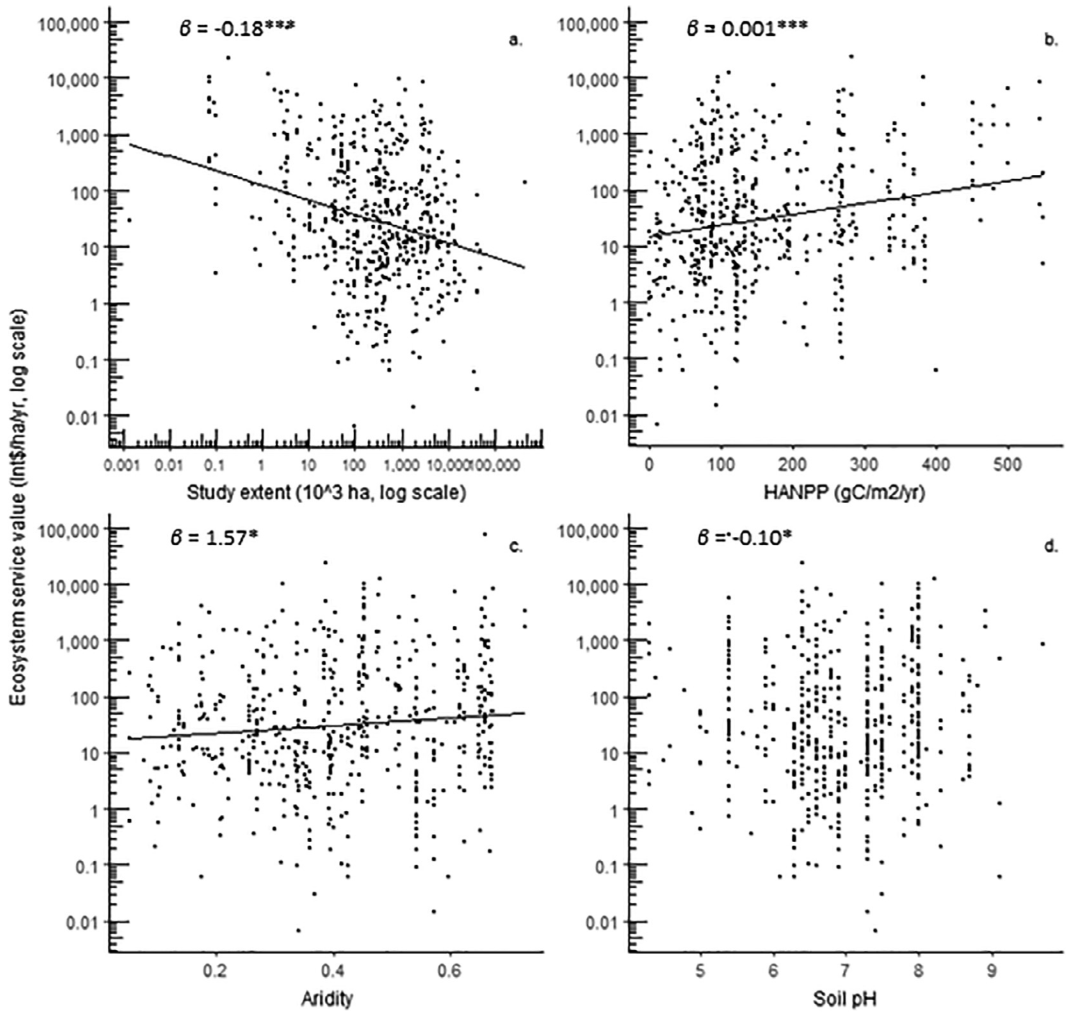

Fig. 3. Linear regressions of dryland ecosystem service value (2007 Int$/ha/yr, log scale) with (a) study extent (10^3 ha, log scale), (b) Human Appropriation of Net Primary Productivity (HANPP, gC/m^2/yr), (c) aridity and (d) soil pH. Standardized regression coefficients (β) are reported, indicating their level of significance with ***, ** and * for 1%, 5% and 10% levels of significance, respectively (using regression model 1 for panels a, c and d, and regression model 2 for panel b).

compared to other dryland ecosystem services. These water-related services appeared also to contain substantial monetary value, when they were compared to monetary values for water-related services from other biomes, which were summarized in a study by de Groot et al. (2012). For instance, fresh water provision had a median value of 138 IntS/ha/yr for the dryland biome in our study, which was higher than for several other biomes, including grassland, temperate forest, tropical forest and inland wetland (ranging between 28–121 Int$/ha/yr), but lower than for some other biomes, including coastal wetland and open water (296 and 1,892 Int$/ha/yr respectively; de Groot et al., 2012). This result is remarkable given the fact that these services had relatively few observations in our study and suggests that more attention needs to be directed towards these type of services in drylands. The high appreciation of water in drylands, which was also illustrated by our finding that the degree of aridity was positively related with dryland value, can be explained by that water is the most limited resource for biological productivity in drylands and therefore highly appreciated (Cuni-Sanchez et al., 2016; Noy-Meir, 1973). As such, this finding highlights the importance of sustainable water management in drylands, especially in view of their essential contribution to safeguarding dryland functioning and the delivery of other dryland services (Bagstad et al., 2012; Le Maitre et al., 2007).

4.2. Local socio-economic and environmental conditions

The high monetary value found for water-related services coincides with a limited impact of local socio-economic and environmental conditions on dryland value, as the degree of aridity, HANPP and soil pH had only a very small effect, while many other indicators, including regional GDP per capita and population density, vegetation productivity and cover, and soil conditions (i.e. fertility, moisture content and crusting), had no effect at all. These results were found despite having collected data for these variables at a local scale; for example using regional GDP data to better reflect welfare conditions in dry regions (MA, 2005; Reynolds et al., 2007). The lack of any effect for regional GDP and population density is particularly remarkable, as in earlier meta-analyses for other biomes, these types of variables were often important in explaining ecosystem service values (Brander et al., 2012; Ghermandi et al., 2010; Quintas-Soriano et al., 2016). For GDP, this contrasting finding may relate to the fact that drylands and their inhabitants are more vulnerable to disturbances than other environments (Verstraete et al., 2009): only a small change in ecosystem service supply may dramatically affect the wellbeing of dryland populations (Christie et al., 2012; O'Farrell et al., 2011). Hence, the exact degree of welfare of the population may not matter for the monetary valuation of this biome's ecosystem services. While this may explain the effects found for regional GDP, the insignificant effects of the other socio-economic and environmental conditions are rather surprising, as

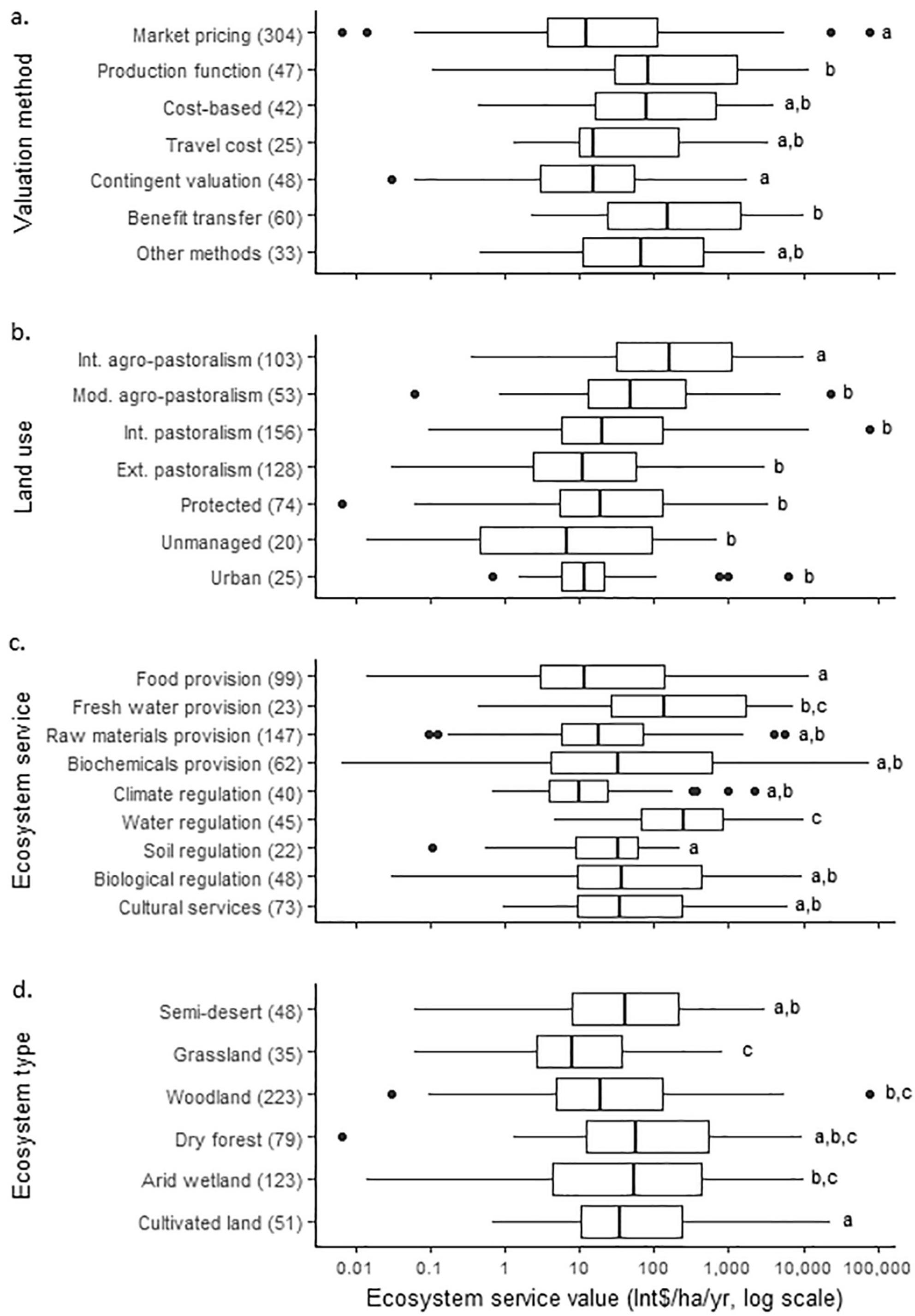

Fig. 4. Boxplots of dryland ecosystem service value (2007 Int$/ha/yr, log scale) with (a) valuation method, (b) land use, (c) ecosystem service and (d) ecosystem type. Post hoc test results are indicated with letter codes next to the boxplots and the number of observations per category are indicated between brackets.

they represented local conditions that are key to dryland functioning, such as soil quality and vegetation productivity. Possibly, this finding indicates that the valuation methods used to estimate the monetary value of dryland ecosystem services have difficulty to capture context-specific conditions of drylands.

4.3. Role of methodological factors

Contrary to our expectations that local socio-economic and environmental conditions would be most important in explaining dryland value, we found that methodological factors, including the type of valuation method and study extent, were important predictors of dryland value. Although previous studies also found evidence that methodological factors were important in predicting ecosystem service value (e.g. Brander et al., 2006; Quintas-Soriano et al., 2016), these studies did not report effect sizes of the significant predictors in their analyses and did not consider such a full range of local socio-economic and environmental conditions as included in our study. While the effect of the selected methodology on the valuation outcome has been suggested multiple times (Brondízio et al., 2010; Martín-López et al., 2014; Spangenberg and Settele, 2010), here we provide quantitative evidence for the dominant impact of such methodological factors on dryland value. This finding is rather alarming, as, ideally, the selected methodology would be merely a means to arrive at an estimated value rather than be dominant in determining the valuation outcome. This indicates that monetary valuation outcomes should be approached with care, as the estimated monetary values for ecosystem services may be overshadowed by potential methodological artefacts introduced by the selected methodology.

For primary valuation methods, the general found trend showed that the production function method estimated higher values than market pricing and contingent valuation methods (although patterns varied per dryland ecosystem service considered, see Schild et al., 2017). Several methods, including market pricing, production function, cost-based and travel cost methods, estimate the actual flow of a service, as they infer their value estimation either directly or indirectly from actual (market) transactions (Bateman et al., 2011; Chee, 2004). Other methods, such as contingent valuation, relate their value estimation to the capacity to deliver a service, as they derive their value estimation from hypothetical transactions (Bateman et al., 2011; Chee, 2004). This may have resulted in the lower value estimates by contingent valuation, although it cannot explain the low monetary values that were found for market pricing. In this case, the low monetary values may have resulted from scarcity: when services are relatively abundant, their market price will be low (Heal, 2000). More generally, these findings indicate that it is important to account for the impact of different primary valuation methods on ecosystem service value estimates.

Furthermore, we found that benefit transfer yielded higher estimated values than other primary valuation methods, which may be due to the secondary nature of this valuation approach. This inherently introduces additional uncertainty in the accuracy of value estimations, such as generalization errors arising from transferring values to unstudied sites (Rosenberger and Stanley, 2006). Our finding that benefit transfer estimated higher monetary values than other methods suggests that this method tends to overestimate the monetary value of dryland ecosystem services, which has been found in previous studies as well (Ready et al., 2004; Rosenberger and Stanley, 2006). The observed overestimation for dryland services suggests that the developed guidelines for the use of benefit transfer to minimize transfer errors may not have been followed that strictly in the practice of dryland valuation (Johnston and Rosenberger, 2010), implying that these dryland values estimated with benefit transfer need to be regarded with caution.

In addition to the large effect of valuation methods, we found that study extent also accounted for a significant part of the variation explained in dryland value. As study extent was negatively related to dryland value, this suggests a decreasing returns to scale relation. This has been found previously in a global analysis for a few specific ecosystem services (Schmidt et al., 2016) and in meta-analyses that were carried out in humid and urban biomes (Brander et al., 2012, 2006; Brander and Koetse, 2011; Enjolras and Boisson, 2008; Ghermandi et al., 2010; Woodward and Wui, 2001). However, while effect sizes are not known in these previous studies, we find here that this effect accounted for a large part of the variation in dryland value. This implies that – as with the choice of valuation method – the selection of the extent of a study area can significantly affect the valuation outcome: the larger the selected study extent, the lower the resulting estimated ecosystem service value may be.

4.4. Complementary effect of land use and ecosystem type

In addition to the large effect of methodological factors, we found that land use explained a relatively large part of variation in dryland value. The monetary value of the most intensively managed land use type was estimated higher than other, less intensively managed, land use types. This finding was also supported by the positive relation found between HANPP and dryland value. The most intensively managed type of land use (i.e. intensive agro-pastoralism) included cropland with large-scale irrigation and pastoral land with high livestock densities. For the management of this type of land use, farmers may have invested more, such as construction costs for irrigation structures, in order to increase the supply of (mostly provisioning) services (Trilleras et al., 2015; van Oudenhoven et al., 2012). As a result, they become more dependent on the production of their land and, hence, appreciate it more than other, less intensively managed types of land use. This effect is illustrated by the trade-off that occurs between roaming pastoralists and sedentary crop farmers, as – in order to convert rangeland into cultivated land – investment in (supplemental) irrigation is needed due to the relative scarcity of water in drylands (Breusers et al., 1998; Franks and Cleaver, 2007).

The effect observed for ecosystem type appeared to be complementary to the effect of land use, which suggests that dryland management and the availability of natural resources are additive to each other. When such a complementary effect indeed exists, this implies that the monetary value derived from dryland ecosystem services can be optimized either by investing in maximizing production of provisioning service(s) from highly managed drylands or by sustainably managing multiple, naturally available ecosystem services from a more naturally managed dryland. As the effect of land use was bigger than that of ecosystem type, the management of dryland resources seems more important for value generation than their natural availability, which is probably due to higher dependence on invested resources in intensively managed drylands. This was confirmed by the finding that ecosystem services that were delivered by more intensively managed, cultivated land types were estimated higher than several other more natural ecosystem types (i.e. grasslands, woodlands and inland wetlands). These findings imply that the existence of intensively managed cultivated drylands optimized for production of provisioning services, next to more sustainably managed drylands providing multiple other services, may be a vital combination to safeguard the flow of ecosystem services in drylands.

5. Conclusion and implications

5.1. Conclusion

The results of this study suggest that the monetary valuation of dryland ecosystem services is only weakly influenced by local socio-economic and environmental dryland conditions. While land use and ecosystem type affected dryland value, other more specific local conditions had only a marginal effect on dryland value. In particular, key dryland conditions, such as those related to local welfare and ecosystem functioning, did not affect dryland value. This suggests that the valuation methods used in current dryland valuation studies do not sufficiently incorporate or cannot adequately capture conditions that are specific to the dryland context. In contrast to the marginal effects of local socio-economic and environmental dryland conditions, the monetary valuation of dryland ecosystem services is heavily influenced by methodological factors. Both the type of valuation method and the study extent greatly affected dryland value. This suggests that the outcome of dryland valuation studies is affected more by the selected methodological approach than by context-specific conditions. These findings have several important implications for future research as well as for policy making.

5.2. Implications for future research

The results of this study indicate that current valuation studies have difficulty to capture context-specific conditions of dryland ecosystem services. In order to improve this, future research should critically evaluate and further develop monetary valuation techniques, as has been pointed out earlier by Braat and de Groot (2012), particularly with regard to better capturing local environmental and socio-economic conditions. In addition, future research could consider additional value types following the newly proposed valuation approach by Pascual et al. (2017). As past dryland valuation studies have focused predominantly on instrumental values, considering relational and intrinsic values may help to more fully capture context-specific conditions. In this respect, the IPBES land degradation assessment would ideally explore whether dryland-specific conditions can be more fully captured this way. Finally, as the value of dryland ecosystem services may be difficult to be captured by monetary valuation approaches alone (Christie et al., 2012; Farley, 2012; Polasky and Segerson, 2009), future research could explore whether complementary, non-monetary valuation approaches, as suggested by Jacobs et al. (2017), can more effectively capture context-specific conditions.

Besides the need for improving monetary valuations techniques, we also recommend future research to improve the reporting of the ecosystem service properties of estimated ecosystem service values, such as the biophysical quantities and properties of the beneficiaries. Current valuation studies often do not report this type of information. Yet, ecosystem service properties have been found to affect valuation outcomes, for instance, in willingness to pay studies (Johnston et al., 2005; Schaafsma et al., 2012). Hence, reporting of ecosystem service properties is relevant. This will not only increase the reliability of individual monetary valuation studies, but will also be valuable to the field as a whole as it can explain contrasting results and contribute to improving monetary valuation of ecosystem services.

5.3. Implications for policy and practice

In addition to implications for future research, our findings also have several important implications for policy and practice. Most importantly, our results show that the monetary valuation of ecosystem services in drylands is still subject to important limitations. Because dryland value is hardly affected by local context-specific conditions but mostly by selected methodology, current dryland valuation may not be able to capture the negative effects of land degradation and the positive effects of sustainable land management. As such, current dryland valuation may not yet be an adequate tool to inform dryland policy and decision making that is aimed at preventing land degradation and promoting sustainable land management.

Yet, if practitioners and decision makers choose to use monetary valuation as a decision support tool, our findings imply that they should not follow monetary valuation blindly. Rather, they need to be aware that methodological choices will inherently affect valuation outcomes. Ways to cope with the observed limitations of monetary valuation include: (1) to select the most relevant spatial scale for the ecosystem service(s) under consideration (for guidelines, see e.g. Hein et al., 2006; Hou et al., 2013), (2) to select the valuation method that is most suitable for the ecosystem service(s) under consideration (for recommendations, see e.g. Bateman et al., 2011; Freeman III, 2003), or – if the choice is ambivalent – to use multiple types of valuation methods so that estimated monetary values can be put in perspective (Boithias et al., 2016), and (3) not to use the same valuation method for the estimation of different types of services, as this may lead to systematic under- or overestimation of values for services for which the method is less suitable.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecoser.2018.06.004.

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