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Differential effects of valuation method and ecosystem type on the monetary valuation of dryland ecosystem services: A quantitative analysis

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ABSTRACT

The method of monetary valuation of ecosystem services has been argued to depend on the type of ecosystem under consideration and the choice of valuation method. Still, the impact of these factors has been hardly studied in a quantitative manner. This study aims to analyze the differential effects of ecosystem type and valuation method on the values estimated for ecosystem services, as well as the potential impact of these effects on aggregated values for ecosystem services. Drylands pose a highly relevant case to investigate these impacts, because they are particularly diverse in ecosystem types, the provided ecosystem services and, hence, are also expected to be estimated with various methods. Our analysis is based on a quantitative analysis of monetary estimates for ecosystem services (expressed in Int$/ha/yr) that were compiled in a comprehensive database containing 512 observations from 57 studies located in drylands worldwide. Our results reveal that the estimated values for dryland ecosystem services depended on the type of ecosystem and method under consideration. Several of these differential effects had a significant impact on the aggregated mean values for dryland ecosystem services. Cultivated lands had high mean values for provisioning services, in particular for food provision, but low values for regulating services. In dry forests, biodiversity-related services were estimated high, in contrast to semi-deserts and arid wetlands. Compared with other methods, market pricing estimated low values for climate regulation and high values for biological regulation. When values were aggregated for ecosystem services, market pricing was found to impact the mean value for climate and biological regulation significantly. Our results highlight the importance of explicit consideration of methods and ecosystem types in monetary valuation, which could lead to more accurate approximation of ecosystem service values.

1. Introduction

The valuation of ecosystem services is a means to express the (relative) importance of the benefits that people obtain from ecosystems (Daily et al., 2009). Although recently more attention is directed towards non-monetary and integrated valuation approaches (Kelemen et al., 2016) and despite various criticisms on monetary valuation approaches (Bockstael et al., 2000; Kallis et al., 2013; Spangenberg and Settele, 2010; Spash, 2008), the empirical studies on the valuation of ecosystem services are still predominantly concerned with economic or monetary valuation of ecosystem services (de Groot et al., 2012; Liu et al., 2010). Also global databases for ecosystem service values, such as The Economics of Ecosystems and Biodiversity (TEEB, 2010a), which are typically used to value ecosystems and management practices, primarily include monetary value estimates.

Meanwhile, it has been observed that monetary valuation of ecosystem services may depend strongly on the appraisal process (Jacobs et al., 2016; Vatn, 2009). The choice of valuation methods has been claimed to direct the valuation outcome (Martín-López et al., 2014; Spangenberg and Settele, 2010; Vatn, 2009; but for a contrast see Brander et al., 2006), also because valuation methods tend to be used outside their originally intended scope of application (Bateman et al., 2011; Farber et al., 2006). In addition, the type

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of ecosystem that is delivering the ecosystem service in question has been noted to affect the monetary value, as the capacity of ecosystems to deliver services may vary based on the underlying functions and processes (La Notte et al., 2015; Villamagna et al., 2013).

However, only a few studies have investigated whether these factors affect the estimated monetary values for ecosystem services in a quantitative manner. Ghermandi et al. (2010) found that the monetary valuation of ecosystem services in wetlands depended on the type of wetland ecosystem considered, while Quintas-Soriano et al. (2016) found that the monetary valuation of ecosystem services in Spain was affected by the methodological approaches of valuation methods.

Yet, although the impact of these factors on the monetary valuation of ecosystem services has been described extensively, still many studies aggregate monetary values of ecosystem services in order to calculate the total economic value of ecosystems or biomes. A well-known example is the study by Costanza et al. (1997) that aggregated values for different ecosystems to arrive at global estimates for the value of nature. More recent examples are studies that have summed up values delivered by different ecosystems to arrive at a total value for a particular study area (e.g. Brenner et al., 2010), while others have aggregated values for ecosystem services that were estimated with different methods and delivered by diverse ecosystems to come to total values for global biomes (e.g. de Groot et al., 2012) or country-wide assessments (e.g. UK National Ecosystem Assessment, 2011).

The extent to and the conditions under which valuation methods and ecosystems affect the monetary values estimated for ecosystem services, and hence also the total economic values, have not been investigated comprehensively and quantitatively so far (Jacobs et al., 2016). Hence, such a quantitative analysis can give important insights into whether these aspects affect the research outcomes of valuation studies. In particular, since the valuation of ecosystem services may be confounded, when different methods or specific ecosystem types are selected preferentially.

The interdependencies between ecosystem service value estimates and the type of ecosystem on the one hand and valuation method on the other hand may, particularly, play a role in drylands, because they include a diversity of ecosystem types within their biome (i.e. as occurring across arid to sub-humid climates, coinciding with a 0.05–0.65 aridity range; Bastin et al., 2017; Maestre et al., 2012; UNCCD, 1994). These ecosystem types include semi-deserts, grasslands, woodlands and dry forests, but also cultivated lands and (semi-)arid wetlands (from here onwards called arid wetlands; Millennium Ecosystem Assessment, 2005; Shackleton et al., 2008). Though the latter category may seem counterintuitive, a high number of arid wetlands occurs within drylands, particularly in semi-arid and sub-humid climate zones (Williams, 1999). These arid wetlands are often temporary due to seasonal or erratic filling (Scoones, 1991; Walker et al., 1995; Williams, 1999). In addition, drylands are diverse in the ecosystem services they can deliver, on which an estimated third of the global human population depends for their well-being and livelihood (Bagstad et al., 2012; Millennium Ecosystem Assessment, 2005; Reynolds et al., 2007; Shackleton et al., 2008). Hence, drylands are a highly relevant case to investigate the possibly confounding, differential effects of ecosystem types and valuation methods on the value estimates of ecosystem services provided.

Our aim was to carry out a systematic analysis of the differential effects of ecosystem type and valuation method on the monetary value estimates (as expressed in Int$/ha/yr) for dryland ecosystem services, based on a quantitative analysis of monetary value estimates for ecosystem services located in drylands worldwide. With differential effects, here, we mean the different effects of dryland ecosystem types and valuation methods on the estimated values for dryland ecosystem services: estimated values for dryland ecosystem services may differ, when they are provided by different dryland ecosystem types or when they are estimated with different valuation methods. In order to address our study aim, we, firstly, aimed to investigate whether and to what extent the monetary value estimates for particular dryland ecosystem services depended on the dryland ecosystem type under consideration. Secondly, this study aimed to analyze whether and to what extent the monetary value estimates for particular dryland ecosystem services depended on the valuation method applied. Thirdly, this study aimed to evaluate the potential impact of specific ecosystem types and valuation methods on the aggregated mean monetary values for dryland ecosystem services in order to assess potential bias when such values are aggregated.

We expected that ecosystem services provided by different dryland ecosystems would have different monetary value estimates, based on the literature cited above. For example, due to the high capacity of arid wetlands to deliver water-related services (i.e. fresh water provision and water regulation), these may be expected to be valued highly. Also, we expected that different valuation methods would lead to different monetary value estimates for the same dryland ecosystem service, as these methods are based on different valuation approaches and address different value types (Bateman et al., 2011; Farber et al., 2006). For example, as market-based methods are specifically developed for valuation of provisioning services, they are expected to provide better estimates for these services than, for example, revealed preference methods which were primarily developed for valuation of cultural services. Finally, we expected that the above-mentioned, differential effects would affect aggregated value estimates for dryland ecosystem services.

2. Methods

2.1. Database of dryland ecosystem service values

We compiled monetary value estimates for dryland ecosystem services in a self-compiled database. As a starting point, we used the TEEB valuation database (van der Ploeg and de Groot, 2010), from which we only extracted studies that were located in drylands, i.e. having a degree of aridity between 0.05 and 0.65 (following the definition of drylands by the UNCCD (1994); thus excluding hyper-arid regions having an aridity lower than 0.05). Based on these records, we went back to the original valuation studies to validate the recorded data and, where needed, recode observations into singular ecosystem service value estimates. Next to the studies extracted from the TEEB database, we complemented the database with valuation studies that were collected from an additional literature review of peer-reviewed and grey literature. Observations were only included in the database when they met the following criteria: (1) the study site was located in a dryland (i.e. having a degree of aridity between 0.05 and 0.65), (2) the recorded value estimate was for a singular ecosystem service, (3) the value estimate for an ecosystem service represented a monetary value that could be standardized, and (4) sufficient data characteristics were available on the ecosystem service, ecosystem type and valuation method. As a result, an observation in our dataset represents the monetary value estimate for a dryland ecosystem service (1) for a specific ecosystem service, (2) delivered by a specific dryland ecosystem, and (3) calculated with a specific valuation method. From some valuation studies, single observations of dryland ecosystem service value estimates were collected, while from other studies multiple observations for dryland ecosystem services value estimates were collected, either for different services or for the same or similar services, that were estimated with different methods or delivered by different ecosystems or study areas. The resulting database contains 512 observations derived from 57 studies (see appendix table A.1 for an overview of these studies).

For each observation of a monetary value estimate of a dryland ecosystem service in the database, we recorded information about (1) the ecosystem service provided, (2) the dryland ecosystem type considered and (3) the valuation method used. Firstly, the ecosystem service of which the monetary value was estimated was defined following the classification for ecosystem services by TEEB (2010b). As some ecosystem services had too few observations to be included individually in the statistical analysis, they were merged with similar services into ecosystem service groups (Table 1). For one specific subservice, we deviated from the TEEB classification to better fit the recorded dryland ecosystem services in our database: TEEB has included the provision of natural extractive products with raw materials provision, however, here, we have included this subservice in the biochemicals provision group, because in drylands these products concern biochemicals, such as natural oils, salts, gums and resins (Gachathi and Eriksen, 2011). In order to examine the impact of clustering ecosystem services into groups, the number of observations, average values and standard deviations were summarized in appendix table A.2. This table showed that the means of the subservices did not differ or when they differed that this was not related to the use of different valuation methods. Hence, clustering subservices into ecosystem service groups created only potentially more within-group variance, but did not lead to statistical artefacts. Together, this resulted in nine dryland ecosystem service groups: (a) provisioning services including food, fresh water, raw materials and biochemicals provision; (b) regulating services including climate, water, soil and biological regulation; and (c) cultural services (Table 1).

Secondly, the dryland ecosystem type that delivered the ecosystem service was specified. We categorized ecosystems into six types, including semi-deserts, grasslands, woodlands, dry forests, arid wetlands and cultivated lands. Semi-deserts ($N = 47$) included open landscapes with low shrub vegetation, such as the succulent and Nama Karoo (i.e. xeric shrublands) and the Masai xeric grass- and shrublands. Grasslands ($N = 35$) consisted of temperate and tropical natural grasslands, including steppes, prairies and rangelands. Woodlands ($N = 218$) included shrublands (i.e. fynbos and Mediterranean shrublands), woodlands (i.e. Mediterranean, Miombo and Acacia woodlands) and savannas (i.e. varying from open to more closed woodlands). Dry forests ($N = 74$) included temperate dry forests and (sub)tropical broadleaf and coniferous dry forests (e.g. tropical dry forests in Ecuador, India and Mexico). Arid wetlands ($N = 106$) consisted of inland wetlands: in addition to a few mangroves, riparian buffers, rivers and lakes, this ecosystem type mainly included seasonal floodplains, swamps and marshes located in sub-Saharan Africa, such as the Waza Lagoon in Cameroon, the Sourou Valley in Burkina Faso and the Okavango Delta in Botswana. Lastly, cultivated land ($N = 32$) included mainly croplands, and a few observations for orchards, greenhouses, aquaculture and urban green spaces.

Thirdly, the valuation method used to estimate the monetary value for dryland ecosystem services was explicitly considered. We grouped the valuation methods that were recorded in our dataset into five valuation approaches based on the TEEB classification (TEEB, 2010c). These methods included: market pricing, production function, cost-based (i.e. avoided cost, replacement cost, and mitigation and restoration cost), travel cost and contingent valuation. In addition, the category ‘benefit transfer’ was created for secondary valuation observations, that were based on one or more primary valuation studies that were adapted to local circumstances. We only included secondary valuation estimates for which double counting with primary valuation observations in the database was ruled out. Finally, the category ‘other methods’ was created for observations that used a valuation method other than the above-defined methods or a combination of above-defined primary methods. A comprehensive review of the different valuation approaches included in our dataset can be found in Bateman et al. (2011), Farber et al. (2006) and Freeman III (2003).

Monetary value estimates calculated for dryland ecosystem services were standardized to 2007 International Dollar per hectare per year (from here onwards called: Int$/ha/yr) in order to have a consistent currency for values that originated from different countries and were estimated for different years. To arrive at 2007 International Dollar per hectare per year values, firstly, we recalculated monetary value estimates that were reported in foreign currencies to their local currency unit using the official exchange rate for the original year of study. Secondly, local currency values were converted to International Dollars using the Purchasing Power Parity (PPP) conversion factor in order to correct for differences in purchasing power between countries. Thirdly, values were standardized to the year 2007 using the GDP deflator in order to correct for price inflation between years. The values for the official exchange rate, PPP conversion factor and GDP deflator were all obtained from World Bank databases (World Bank, 2010).

2.2. Statistical analysis

In the statistical analysis, the dependent variable was the monetary value for dryland ecosystem services. As the data for the dependent variable did not follow a normal distribution, we transformed it using its logarithm ($^{10}\log$) in order to be able to run

Table 1
Dryland ecosystem service groups in the dryland database ($N = 512$), including a description of the specific services included and their number of observations.

Ecosystem service class	Dryland ecosystem service group[a]	Description	Number of observations
Provisioning	Food provision	Fish, meat (i.e. wildlife and livestock), vegetables and forest products (i.e. honey and fruit)	97
	Fresh water provision	Drinking, irrigation and industrial water	21
	Raw materials provision	Bulk materials, including fuelwood, charcoal, fibers (i.e. thatch, reeds and grasses), timber and fodder	142
	Biochemicals provision	Genetic and medicinal resources (i.e. medicinal plants and bioprospecting), ornamental resources (i.e. decorations and handicrafts), forest products (i.e. cork and gum) and other natural extractive products (i.e. natural oils, salts, dyes)	60
Regulating	Climate regulation	Carbon sequestration	21
	Water regulation	Water flow regulation, water purification and flood attenuation	38
	Soil regulation	Soil erosion prevention and maintenance of soil fertility (i.e. nutrient deposition and cycling)	22
	Biological regulation	Biological control, pollination, and maintenance of biological and genetic diversity	45
Cultural	Cultural services	Recreation, (eco)tourism, hunting, aesthetic and inspirational services	66

[a] Following the TEEB classification for ecosystem services (TEEB, 2010b).

parametric tests in the subsequent statistical analysis. After the ^{10}log transformation, the dependent variable followed a normal distribution, which was tested using the Shapiro-Wilk test ($W = 0.99$, $p = 0.16$).

In order to address our research aims, we carried out two statistical analyses. First, we defined two interaction terms for (1) ecosystem service with ecosystem type and (2) ecosystem service with valuation method. We tested whether these interaction terms were significant in two separate two-way ANOVAs (at $p < 0.05$ level of significance). To understand the combinations of (1) ecosystem services with ecosystem type and of (2) ecosystem services with valuation method that contributed to the significant interaction terms, we calculated the mean values for each of these combinations. Using a one-way ANOVA, we tested whether these means differed significantly from each other (at $p < 0.05$ level of significance). Subsequently, we tested which specific combinations differed significantly from each other using the Tukey post-hoc test (at $p < 0.05$ level of significance). For this latter analysis, combinations having only one observation were excluded from the dataset (this concerns seven combinations; see appendix table A.3 and A.4).

Second, in order to evaluate the impact of not accounting for different valuation methods and ecosystem types, we calculated the overall mean value for each dryland ecosystem service based on the database (N = 512). In order to evaluate the impact of aggregating estimated values across dryland ecosystems and methods, we analyzed whether the overall mean values for dryland ecosystem services changed when specific categories or combinations were omitted as compared to the overall aggregated values. For ecosystem types, omitted categories were selected based on the results of the differential impacts of ecosystem types on the monetary value estimates of dryland ecosystem services. For valuation methods, a category was created that excluded benefit transfer, which is a secondary valuation method, and 'other methods', which constituted diverse methodological approaches that did not fit within one of the specified categories. As valuation methods may have been used to estimate values for a wider range of services than for which they were primarily developed (Bateman et al., 2011; Farber et al., 2006; Freeman III, 2003), another category was created that only included the combinations of methods with the ecosystem services for which they were designed originally (see appendix table A.5 for an overview). In order to evaluate whether the differences among dryland ecosystem services changed as compared to the overall aggregated value estimates for dryland services, we tested for differences among the mean aggregated values for dryland services within these newly created categories using the one-way Anova test and for multiple comparisons among dryland services using the Tukey post-hoc test (both at $p < 0.05$ level of significance).

3. Results

3.1. Description of observations in the dryland database

More than half of the observations in the dataset were located in Africa (Fig. 1). A substantial number also came from Europe and Asia, while North America, South America and Australia had only a few observations. Nearly all combinations of dryland ecosystem services with dryland ecosystem types were present in the dataset, except for semi-desert, which lacked observations for food, fresh water and biochemicals provision, and climate and soil regulation services. These latter ecosystem services may either not or to a lesser extent be provided by semi-deserts or be lacking in the valuation studies that were collected in the database. The number of observations varied greatly over the different combinations, ranging from only one observation for seven combinations up to $N = 71$ for raw materials provision from woodlands (see appendix table A.3).

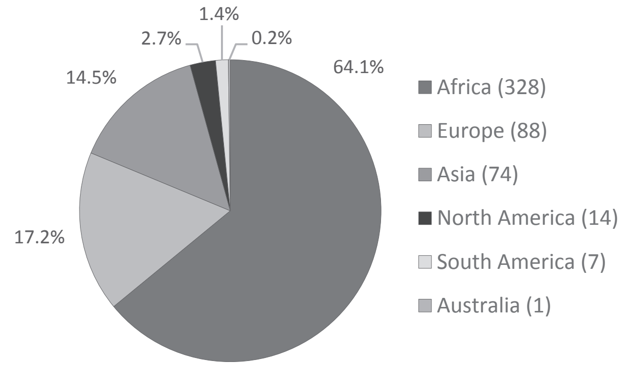

Fig. 1. Number of observations on each continent in the dryland database (N = 512) indicated as a percentage (%) in the diagram and their actual number of observations is given between brackets.

For valuation methods, observations for 39 out of a potential of 63 combinations of dryland ecosystem services and valuation methods were present in the dataset. Most of the valuation methods, including market pricing, production function, cost-based and benefit transfer methods, had observations for most ecosystem services. The other valuation methods, including travel cost, contingent valuation and other methods, had only observations for a few services. Specifically, the travel cost method had only observations for cultural services. Furthermore, large variation was found in the number of observations per combination of dryland ecosystem service and valuation method, ranging from one observation for several combinations to $N = 90$ for food provision and $N = 129$ for raw materials provision, both estimated with the market pricing method (appendix table A.4).

3.2. Differential effects of ecosystem type

The interaction term defined for the combinations between ecosystem services and ecosystem types was found to be highly significant ($F(41,463) = 4.52$, $p < 0.001$), which showed that dryland ecosystem services have different monetary value estimates when they are provided by different dryland ecosystems, which was according to expectations. The mean estimated values for specific ecosystem services provided by different dryland ecosystems varied widely: from less than 1 to over 3000 Int$/ha/yr (Fig. 2a and appendix table A.3). Fig. 2a shows that no homogenous pattern of mean value estimates existed across dryland ecosystem types and ecosystem services. Notably, cultivated lands had relatively high mean values for provisioning services and low mean values for regulating services, as compared to the other dryland ecosystem types. Arid wetlands received relatively high mean values for regulating services (except for biological regulation) as compared to the other dryland ecosystems. For biological regulation, dry forests had relatively a high mean value, while semi-deserts had a remarkably low mean value. Apart from a few exceptions, semi-deserts, grasslands and woodlands had relatively low mean values for all services as compared to other ecosystem types.

The post-hoc analysis showed that nine different groups of ecosystem service and ecosystem type combinations could be distinguished (Table 2), in which group I had significantly lower monetary value estimates than group IX. This result showed that mean value estimates for the combinations in group IX, including fresh water provision and water regulation by arid wetlands, water

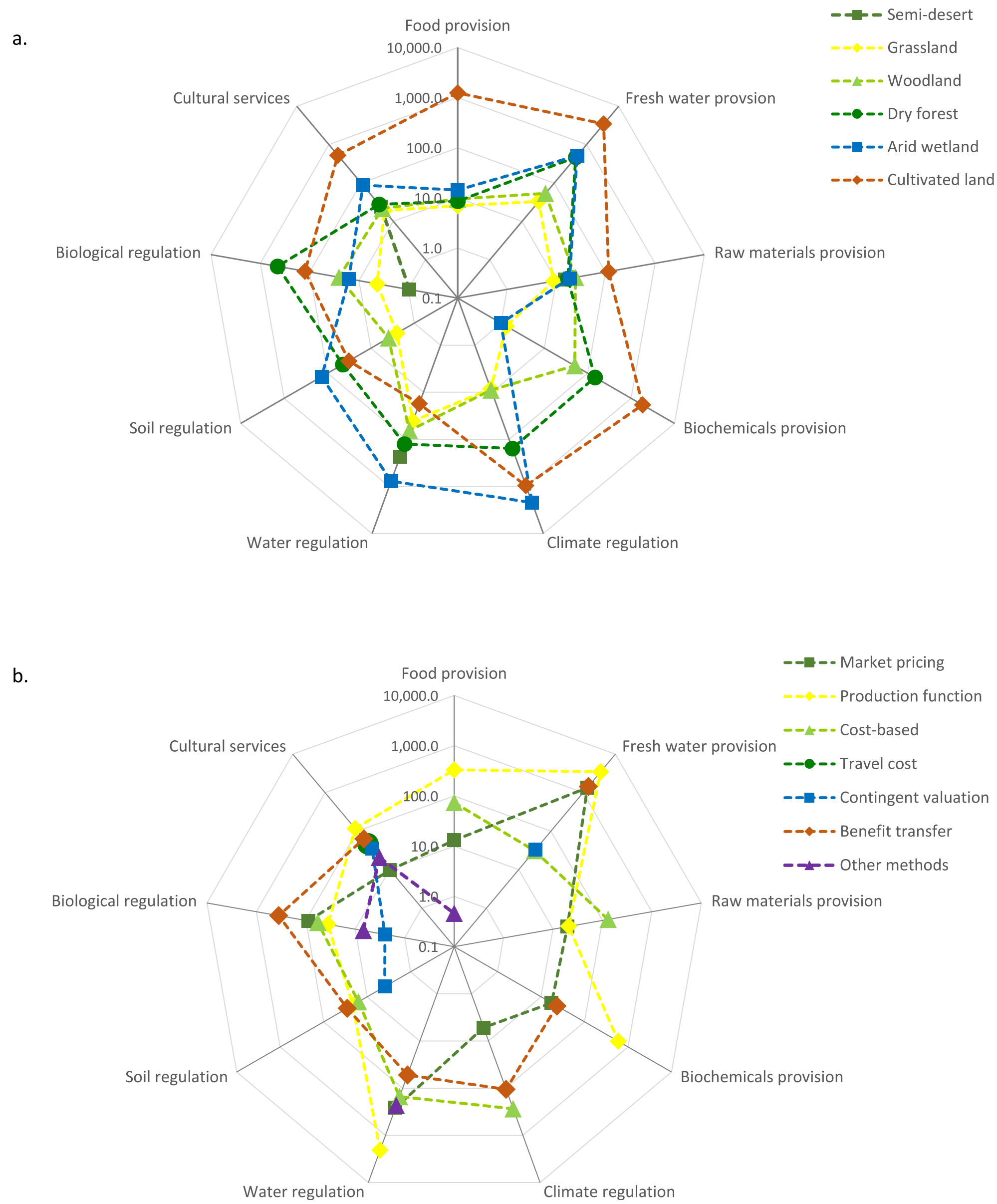

Fig. 2. Radar plots showing the mean monetary value estimates of the combinations of dryland ecosystem services (expressed in Int$/ha/yr, on a log scale and indicated on the nine radar axes) and (a) dryland ecosystem types and (b) valuation methods (both displayed on the radar axes using different colors). Mean value estimates represent the back-transformed ^{10}log mean values (using their exponential) and are based on the dryland database (N = 512). Numeric values of the mean value estimates for all combinations can be found in appendix tables A.3 and A.4. To increase visibility dots are connected with punctuated lines, though these lines themselves are meaningless.

regulation in semi-deserts, food provision from cultivated lands, and biochemicals provision and biological regulation by dry forests, were significantly higher than mean value estimates for combinations in group I, including food and biochemicals provision by arid wetlands, food provision by woodlands, soil regulation in grasslands, and biological regulation in semi-deserts. The number of observations for the combinations in these groups varied considerable (N = 5–43; appendix table A.3).

Also, these findings showed specific differences that occur within the same ecosystem service and the same dryland ecosystem type. Significant differences within an ecosystem type were found for semi-deserts, where water regulation had significant higher mean value estimates than biological regulation, and for arid wetlands, where fresh water provision and water regulation had higher mean value estimates than food and biochemicals provision. Significant differences within ecosystem services were exemplified by food provision being estimated significantly higher in cultivated lands than in woodlands and arid wetlands. Also, biochemicals provision from dry forests was estimated significantly higher than from arid wetlands. Furthermore, biological regulation was estimated significantly higher in dry forests than in semi-deserts.

3.3. Differential effects of valuation method

The interaction term between dryland ecosystem services and valuation methods was highly significant ($F(31{,}473) = 4.57$, $p < 0.001$), which showed that specific methods estimate the value of specific dryland ecosystem services differently, as expected. In Fig. 2b, the mean monetary value estimates for each dryland ecosystem service per different valuation methods are depicted (see appendix table A.4 for the mean values and standard

Table 2
Multiple comparisons of the combinations of dryland ecosystem services with dryland ecosystem types, indicating to which group each combination belongs (in roman numbers) as tested with the Tukey post-hoc test, in which combinations that showed the same behavior belonged to the same group.[a] Combinations in group I (having lowest mean monetary value estimates) differed significantly from those in group IX (having highest mean monetary value estimates; at $p < 0.05$ levels of significance). Both these groups are indicated with bold symbols.[b]

Dryland ecosystem service	Dryland ecosystem type					
	Semi-desert	Grassland	Woodland	Dry forest	Arid wetland	Cultivated land
Food provision		III	**I**	IV	**I**	**IX**
Fresh water provision		V	V		**IX**	VIII
Raw materials provision	II	V	VI	III	III	
Biochemicals provision		V	VII	**IX**	**I**	VIII
Climate regulation		V	II	V		
Water regulation	**IX**	V	V		**IX**	
Soil regulation		**I**		V	V	V
Biological regulation	**I**	V	V	**IX**	V	V
Cultural services	V	V	III	V	VII	VII

[a] The combinations between ecosystem services and ecosystem types were tested whether their means were significantly different from each other using the Tukey post-hoc test. Combinations that had the same differences in comparison to other combinations were grouped together, as indicated with roman numbers.
[b] The intermediate groups II-VIII overlap in varying degrees with each other: this is depicted in appendix figure A.1.

deviations). This figure reflects the heterogeneity in mean value estimates across dryland ecosystem services and valuation methods. The amount of variation depended on the ecosystem service and valuation method considered, as, for example, little variation was observed in the mean estimated values for cultural services, but much variation was observed for food provision and biological regulation. In general, values that were estimated with benefit transfer and production function were on the higher value end, while market pricing was on the lower end. The category ‘other methods’ showed a very variable pattern in mean value estimates for different dryland services.

In the multiple comparison analysis, four different groups were found (at $p < 0.05$ level of significance; Table 3). The combinations included in group I differed significantly from those in group IV, in which group I had significantly lower value estimates than group IV. This showed that the mean value estimates for the combinations of fresh water provision estimated with either market pricing, production function or benefit transfer methods were significantly higher than the mean value estimates for biological regulation estimated with the contingent valuation and most of the services estimated with market pricing (i.e. food, raw materials and biochemicals provision and climate regulation). While all interactions in group I were based on a considerable number of observations ($N = 12–129$; appendix table A.4), the combinations occurring in group IV should be interpreted with care because they had a low number of observations ($N = 2–3$). Apart from soil regulation and cultural services, all services showed strong variation in mean value estimates depending on which valuation method had been used. Across valuation methods, some methods, including cost-based methods, showed little variation among mean value estimates for different services, while other valuation methods, including market pricing, production function and benefit transfer, showed considerable variation across services. Particularly notable results here were the high value estimates for fresh water provision that were estimated with market pricing, production function and benefit transfer methods. Also, the low estimated values for climate regulation estimated with market pricing and for biological regulation estimated with contingent valuation stood out.

3.4. Impacts of differential effects on aggregated values

In order to evaluate the impact of not specifically accounting for valuation method or ecosystem type when aggregating the monetary value estimates for dryland ecosystem services, we aggregated the value estimates within our dataset into the overall mean monetary values for dryland ecosystem services. These overall mean values for dryland ecosystem services differed significantly from each other ($F(8,503) = 5.00$, $p < 0.001$). Fig. 3 shows the overall estimated mean values for the different ecosystem services provided by drylands. Overall, estimated mean values for water-related services, including fresh water provision and water regulation, were high, which have been analyzed in detail in Schild et al. (in review). Post-hoc test results showed that the mean value estimates for fresh water provision and water regulation were

Table 3
Multiple comparisons of the combinations of dryland ecosystem services with valuation methods, indicating to which group each combination belongs (in roman numbers) as tested with the Tukey post-hoc test, in which combinations that showed the same behavior belonged to the same group.[a] Combinations in group I (having lowest mean monetary value estimates) differed significantly from those in group IV (having highest mean monetary value estimates; at $p < 0.05$ levels of significance). Both these groups are indicated with bold symbols.[b]

Dryland ecosystem service	Valuation method						
	Market pricing	Production function	Cost-based	Travel cost	Contingent valuation	Benefit transfer	Other methods
Food provision	**I**	III					
Fresh water provision	**IV**	**IV**	II		II	**IV**	
Raw materials provision	**I**	II	II				
Biochemicals provision	**I**	III					
Climate regulation	**I**		III			II	
Water regulation			II			II	III
Soil regulation		II	II				
Biological regulation	II	II	II		**I**	III	II
Cultural services	II			II	II	II	II

[a] Same as in Table 2.
[b] The intermediate groups II and III overlap in varying degrees with each other: this is depicted in appendix figure A.2.

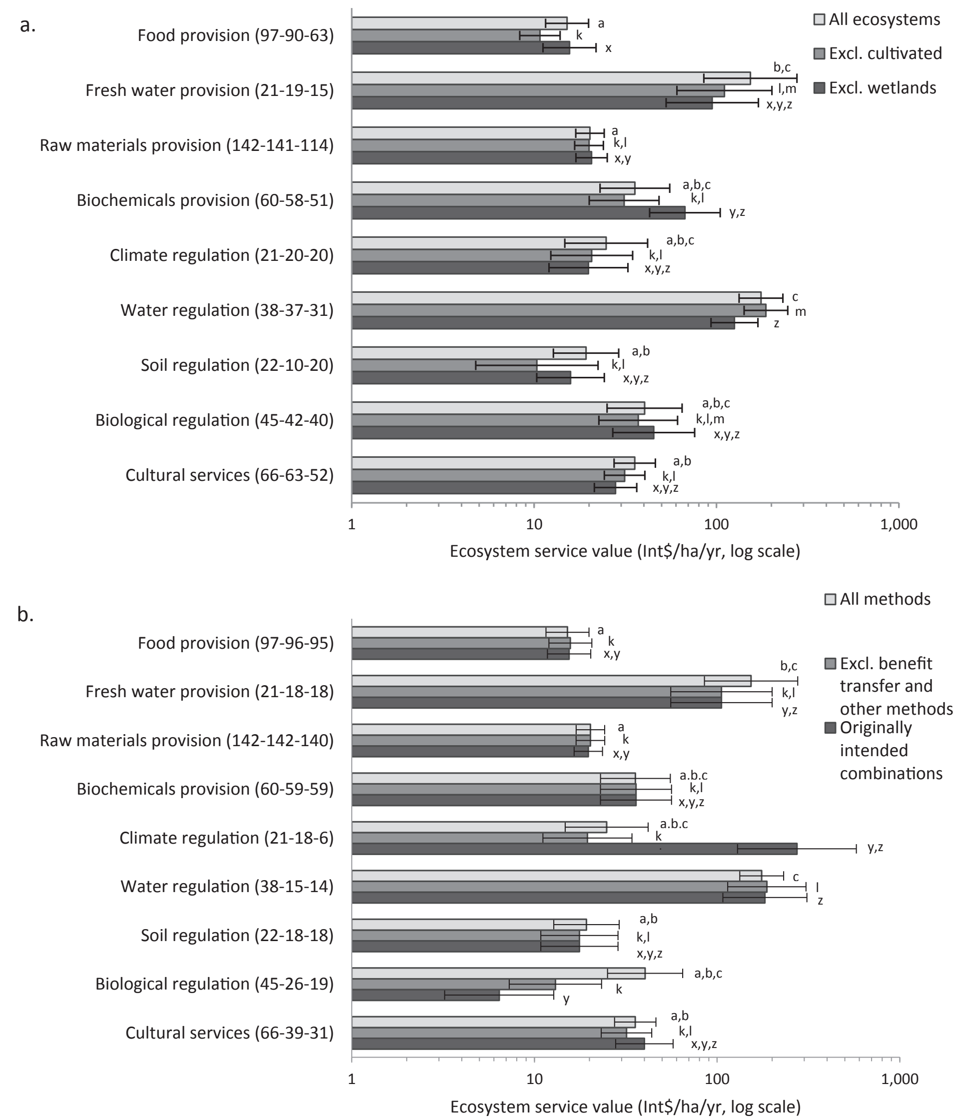

Fig. 3. Aggregated mean monetary values for dryland ecosystem services (expressed in Int$/ha/yr, on a log scale), showing in panel (a) all ecosystem types, ecosystems excluding cultivated lands and ecosystems excluding arid wetlands, and in panel (b) all valuation methods, methods excluding benefit transfer and 'other methods', and 'originally intended combinations' including only the methods with ecosystem services combinations for which they were primarily developed for (see table A.5 in the appendix). Mean values represent the back-transformed ^{10}log mean values (using their exponential) based on the dryland database (N = 512), error bars indicate ± 1 standard error of the mean and post-hoc test results are indicated with the letter codes next to each bar. The number of observations for each ecosystem service is shown in parentheses on the y-axis for each bar category, respectively.

significantly higher than for food and raw materials provision. In addition, water regulation had a significantly higher mean value estimate than soil regulation and cultural services in the post-hoc test.

The ranking in the monetary value estimates for dryland ecosystem services was found to strongly depend on particular combinations of ecosystem services with ecosystem types and ecosystem services with valuation methods. In order to evaluate the impact of specific ecosystem types on the aggregated monetary value estimates for dryland ecosystem services, we excluded two ecosystem types from our dataset that were expected to impact the mean value estimates. First, we excluded cultivated lands, as this ecosystem showed a contrasting pattern having relatively higher estimated values for provisioning and cultural services and relatively lower estimated values for regulating services as compared to all other ecosystem types (see Fig. 1a). In particular, food provision was found to be significantly higher in cultivated lands than in several other dryland ecosystems (Table 2). When cultivated lands were excluded from the dataset ($N = 480$), mean value estimates for dryland ecosystem services were still significantly different from each other ($F(8,471) = 5.79$, $p < 0.001$; Fig. 3a) and also the ranking was hardly affected according to the post-hoc test results. The only difference was that fresh water provision was no longer estimated significantly higher than raw materials provision, but water regulation was estimated significantly higher than two more services, being biochemicals provision and climate regulation.

Second, arid wetlands were excluded from ecosystem types, as this ecosystem is relatively 'wet' in comparison to the otherwise dry ecosystems that are part of drylands and because it had significantly higher mean value estimates for water provisioning and regulating services. When arid wetlands were excluded from the dataset ($N = 406$), mean value estimates for dryland ecosystem services differed significantly from each other as well ($F(8,379) = 3.71$, $p < 0.001$). When comparing the ranking for 'all ecosystem types' and 'wetlands excluded' (Fig. 2a), on the one hand fresh water provision was no longer estimated significantly higher than food and raw materials provision and water regulation no longer higher than soil regulation and cultural services, though, on the other hand, biochemicals provision was estimated significantly higher than food provision. This latter finding demonstrates how low- or high-end value estimates for a particular services generated by a specific ecosystem type affected overall aggregated values.

To evaluate the impact of specific valuation methods or combinations of specific methods and services on the aggregated mean value estimates for dryland services, we analyzed how different selections of methods and combinations affected the aggregated values in two different ways. First, we analyzed the impact of omitting benefit transfer and 'other methods'. When they were excluded from the dataset ($N = 431$), mean value estimates for dryland ecosystem services still differed significantly from each other ($F(8{,}422) = 2.89$, $p = 0.004$). Post-hoc test results showed that on the one hand water regulation was no longer estimated significantly higher than soil regulation and cultural services, but on the other hand water regulation was estimated higher than biological regulation (Fig. 3b). In particular, a notable decrease in the aggregated mean value estimate for biological regulation was observed when benefit transfer and 'other methods' were excluded. Apart from this specific effect, however, the exclusion of benefit transfer appeared only to have a small effect on aggregated mean value estimates, showing that this category - which indirectly included a combination of primary methods - did not lead to any artificial effects in the results.

Second, only combinations were included for which valuation methods were originally developed for (see appendix table A.5). Mean value estimates for dryland ecosystem services in this dataset ($N = 400$) differed significantly from each other as well ($F(8{,}391) = 4.04$, $p < 0.001$). Post-hoc test results showed that − in contrast to the situation when all methods were aggregated − biological regulation had significantly lower value estimates than fresh water provision, climate and water regulation (Fig. 3b). Again, the aggregated mean value estimate for biological regulation decreased: this time due to the exclusion of market pricing. Even more notable was the dramatic increase in the aggregated mean value estimate for climate regulation when market pricing was excluded.

4. Discussion

This study aimed to analyze the differential effects of ecosystem type and valuation method on the value estimates for dryland ecosystem services. We find that dryland ecosystem service value estimates depended on the ecosystem type and valuation method under consideration.

4.1. Dependence on ecosystem type

Our analysis supported our expectation that the estimated values for dryland ecosystem services depend on the type of ecosystem that delivered these services. Several specific combinations of ecosystem types and ecosystem services stood out. We found that provisioning services, and in particular food provision, from cultivated drylands were valued highly. In our dataset, food provision value estimates were mainly concerned with crop production, which may explain the high value found in cultivated drylands: croplands are often specifically managed for food production and principally aimed at achieving high yields (Power, 2010). Such intensive land use may crowd out the provision of other services, which may also explain why regulating services were valued much lower than provisioning services in cultivated drylands. The low values for regulating services compared to provisioning services are alarming, as regulating services, such as water infiltration, soil fertility and pollination, are essential to maintain provisioning services in the long run (Gordon et al., 2010; Power, 2010). As population growth and increasing food demand in drylands are expected to drive expansion and intensification of dryland cultivation (Stringer, 2009), this calls for stimulating a fuller appreciation by dryland farmers and decision makers of the importance of these regulating services in sustaining food provisioning in drylands.

Furthermore, biodiversity-related services, including biochemicals provision and biological regulation, were perceived particularly high in dry forests as compared to other dryland ecosystem types. Dry forests may have, in comparison to other dryland ecosystems, a high capacity to deliver such services, as they are characterized by a rich biodiversity (Miles et al., 2006) and are well represented among the global biodiversity hotspots (Myers et al., 2000). In our dataset, biochemicals provision in dry forests included predominantly bioprospecting for medicinal substances. The high value estimates for this service may be explained by the considerable interest of pharmaceutical companies and society in general that comes along with the use of these materials in manufacturing and developing (new) medicines (Gundimeda et al., 2006). The value estimates for biological regulation in dry forests included mainly maintenance of species and biodiversity, which were predominantly estimated based on willingness to pay, either directly using contingent valuation or indirectly using benefit transfer based on willingness to pay values. This finding suggests that people may perceive the maintenance of biodiversity in dry forests as highly important, which underlines the importance of safeguarding the provision of these biodiversity-related services when managing dry forests, in particular given that the remainder of dry forests is threatened by loss and degradation (Miles et al., 2006).

In addition to the dependence on ecosystem type found at the high value end, we also found dependencies for several mean value estimates for ecosystem services that were provided by semi-deserts, grasslands and woodlands that were at the lower value end. For instance, biological regulation was estimated the lowest in semi-desert, while being estimated the highest in dry forest, and food provision was estimated the lowest in woodland, while estimated the highest in cultivated land. These low estimates may be due to that these ecosystem types may deliver these services in a lower amount, different form or lesser quality, as these ecosystems generally have a lower primary productivity (Noy-Meir, 1973). Yet, it is important to keep in mind that even though the estimated monetary value for a service may be low, the service could be vital for the subsistence of local populations. Monetary valuation may not fully capture such a crucial social value (O'Farrell et al., 2011). To better capture such values, it may be helpful to use non-monetary valuation techniques in addition to monetary valuation tools (Kelemen et al., 2016) in order to avoid the risk that these potentially low values might lead to further marginalization in public opinion and decision making, as drylands are already perceived as marginal lands (Reynolds et al., 2007).

In conclusion, the dependencies of dryland ecosystem service values on specific dryland ecosystem types showed that services were valued differently in different ecosystems, which appeared, for instance, to be due to their type of management (as for food provisioning services by cultivated land) or their high capacity to deliver specific services (as for biodiversity-related services by dry forest). Despite the broadness of the categories in which we had pooled our data, variation within the categories did not dominate the results, as we found a substantial number of differential effects among specific ecosystem services and ecosystem types. These findings indicate that explicit consideration of the specific type of dryland ecosystem is key in dryland ecosystem services valuation in order to account for these dependencies.

4.2. Dependence on valuation method

Our second expectation, that dryland ecosystem service value estimates depend on the method used, was supported by our findings as well. We found such dependence for several specific combinations of methods and services. For biological regulation, we found that especially contingent valuation estimated low values in comparison to other combinations. In our dataset, all value estimates for biological regulation with contingent valuation concerned non-use values (i.e. option, bequest and existence values) for the maintenance of genetic and biological diversity. As these types of values and services are less tangible (Bateman et al., 2011), people may have had difficulty to grasp the value of biological regulation, because they may find it difficult to understand its meaning and to comprehend its importance. In contrast, more tangible services, such as fresh water provision (i.e. direct water supply) and cultural services (i.e. dominated by recreation and tourism, such as wildlife viewing) were estimated consistently higher with contingent valuation. In order to better capture the different value dimensions of biological regulation, it could be useful to use an integrated approach in which non-monetary and monetary valuation approaches are combined (Jacobs et al., 2016; Kelemen et al., 2016). This could be of particular relevance for drylands, as they are predominantly located in less developed regions (Reynolds et al., 2007), where monetization of values is a less common practice (Christie et al., 2012).

While biological regulation was estimated relatively low when contingent valuation methods were used, we found that this service was estimated high by the market pricing and benefit transfer methods. This may relate to the fact that these market prices, which mainly concerned the net revenue of maintenance of a nursery habitat for fish species and alternative options for biodiversity conservation, were net values that were corrected for the costs of production. Hence, they may not have been corrected for market distortions, such as taxes or subsidies (Bateman et al., 2011). In case of benefit transfer, the nature of this secondary valuation method may have led to systematically higher value estimates here, because the values were derived elsewhere (e.g. Brouwer, 2000).

Next to method dependencies for biological regulation, we also found a distinct impact of market pricing on the value estimate for climate regulation (i.e. carbon sequestration), which estimated very low values compared to other methods. This may be related to that most observations in our dataset used a carbon price of 20 $/tC (for 1991–2000 period), which appears to incorporate only part of the social costs that are involved in carbon, such as temperature rises, increases in precipitation levels, sea level rises and increases in the occurrence of extreme events, such as droughts and floods. A best estimate for these social costs has been estimated at 46 $/tC for the year 2000 (with a 23–92 $/tC sensitivity range, at 2000 prices), which is assumed to increase with time (Clarkson and Deyes, 2002). The market prices in our database may be lower than this optimal price, because the market for carbon is known to be very vulnerable to market failures, such as illustrated by the information problems and misuse of market power in the European Union emissions trading scheme (Andrew, 2008).

The finding that market pricing estimated climate and biological regulation consistently lower than methods that are considered more appropriate for their valuation (i.e. production function and cost-based methods; Bateman et al., 2011; Farber et al., 2006), suggests that market pricing, although proven to be a valuable tool for the valuation of provisioning services (Bateman et al., 2011), may be less adequate in capturing values of regulating services. It has been argued previously that market pricing for other than provisioning services can be easily prone to errors, as it would attempt to estimate a price for non-existent market impacts, as these services are usually not directly traded in markets (Daily et al., 2000). Here, we find empirical evidence to underpin these theoretical arguments, which implies that market pricing may be better avoided for the valuation of regulating services.

Lastly, we also observed some method dependence in the valuation of fresh water provision: market pricing, production function and benefit transfer methods estimated the value of this service substantially higher than other types of methods. Fresh water provision, which included water supply for domestic, agricultural and industrial use, is a limited resource in dry areas (Noy-Meir, 1973). Hence, methods that base their valuation on the market – which values scarce goods higher than abundant ones – may lead to high prices for water, either directly through the water price (i.e. market pricing method) or indirectly through its input in dryland agricultural production (i.e. production function method). The use of the benefit transfer method may introduce additional uncertainties due to its secondary valuation nature, which may have led to high value estimates here. As benefit transfer also estimated a high mean value for biological regulation, these high values may be either due to methodological bias of benefit transfer or be inherent to valuation of these specific dryland services with this method. Yet, we observed these impacts of benefit transfer only for these two ecosystem services, suggesting that the impact of this method on value estimates was not as dramatic as would have been expected (Brouwer, 2000).

In conclusion, we found that the mean value estimates for particular ecosystem services depended on the type of method, either because they appeared to have difficulty to grasp their value or to be outside their methodological scope. Moreover, the use of a less suitable method had a considerable impact on aggregated values for dryland ecosystem services. The differential effects of methods and ecosystem service were not dominated by the variation in method and ecosystem service categories given that we found a substantial number of differential effects. These findings imply that methods need to be considered explicitly in dryland valuation studies.

4.3. Implications for valuation

This study provides the first quantitative evidence of differential effects, showing that the valuation of dryland ecosystem services depended on ecosystem type and valuation method. Previous literature has argued extensively that valuation methods are expected to affect valuation outcomes (Martín-López et al., 2014; Spangenberg and Settele, 2010; Vatn, 2009), but this has only been sparsely substantiated with empirical evidence (Quintas-Soriano et al., 2016).

The findings in this study have several implications for future research. First, the finding that some methods have a dominant impact on estimated ecosystem service values in drylands implies that when valuing ecosystem services, the suitability of a method for a valuation exercise needs to have priority over other considerations, such as the time- or cost-effectiveness of methods.

Second, our findings imply that the estimated values for dryland ecosystem services cannot be simply aggregated for drylands. Such aggregation neglects the interdependencies between ecosystem services, ecosystem types and methods and obscures the underlying variation. Moreover, it may bias the result as we found that some low- or high-end estimates were dominating overall aggregated values. In this study, we, therefore, abstained from reporting any grand, overall aggregated value for drylands, despite the increasing tendency to do so (e.g. de Groot et al., 2012; UK National Ecosystem Assessment, 2011). We advise other scholars to be careful in this respect as well.

Third, our results may have implications for monetary valuation within other biomes, as the observed differential effects of methods and ecosystem types can play a role here as well. Our results indicate that it is essential to explicitly account for the type of ecosystem and valuation method in both primary and secondary valuation studies. In primary valuation studies for instance, the explicit consideration of different (sub)ecosystem types is necessary to account for any differences among ecosystems. Such observations may also apply to other biomes.

Finally, the findings of our study may also have implications for studies that aim to estimate the total economic value of specific areas based on aggregating values across ecosystem services. As we found a distinct impact of the differential effects of ecosystem types and methods on the aggregated values for dryland ecosystem services, these differential effects may also play a role when values are aggregated for other biomes or localities, such as local study areas, countries or regions. As such, these type of studies need to explicitly account for the impact of differential effects on aggregated values.

5. Conclusions

Our study showed that monetary value estimates for dryland ecosystem services depended strongly on the ecosystem type and method considered. The patterns and extent of the impact of these differential effects differed per ecosystem service, ecosystem type and method concerned. We show that these differential effects impact values when they are aggregated across methods and ecosystem types. As no study has yet assessed these differential effects of ecosystem types and valuation methods on ecosystem service values in a comprehensive and quantitative way, this study provides the first empirical evidence that ecosystem types and method affect monetary estimates for dryland ecosystem service values. When these factors are taken into account, the accuracy of the approximation of ecosystem service values can be substantially improved, which may in turn lead to more meaningful information to feed policy and decision making with regard to dryland management.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.jaridenv.2017.09.001

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