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Prioritizing West African medicinal plants for conservation and sustainable extraction studies based on market surveys and species distribution models

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ABSTRACT

Sub-Saharan African human populations rely heavily on wild-harvested medicinal plants for their health. The trade in herbal medicine provides an income for many West African people, but little is known about the effects of commercial extraction on wild plant populations. Detailed distribution maps are lacking for even the most commonly traded species. Here we combine quantitative market surveys in Ghana and Benin with species distribution models (SDMs) to assess potential species' vulnerability to overharvesting and to prioritize areas for sustainable extraction studies. We provide the first detailed distribution maps for 12 commercially extracted medicinal plants in West Africa. We suggest an IUCN threat status for four forest species that were not previously assessed (*Sphenocentrum jollyanum*, *Okoubaka aubrevillei*, *Entada gigas* and *Piper guineense*), which have narrow distributions in West Africa and are extensively commercialized. As SDMs estimate the extent of suitable abiotic habitat conditions rather than population size per se, their output is of limited use to assess vulnerability for overharvesting of widely distributed species. Examples of such species are *Khaya senegalensis* and *Securidaca longipedunculata*, two trees that were reported by market vendors as becoming increasingly scarce in the wild. Field surveys should start in predicted suitable habitats closest to urban areas and main roads, as commercial extraction likely occurs at the shortest cost distance to the markets. Our study provides an example of applying SDMs to conservation assessments aiming to safeguard provisioning ecosystems.

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1. Introduction

Millions of African people rely on medicinal plants for their primary health care (Antwi-Baffour et al., 2014). The trade in herbal medicine is of considerable economic value, providing income for large numbers of people involved in collecting, processing, transport and sale of plants, even though most activities take place in the informal economic sector (Cunningham, 2001; Sunderland and Ndoye, 2004; Williams et al., 2007; Laird et al., 2010). The bulk of the herbal medicine in Africa is harvested from the wild, and there are indications that several species in high demand suffer from overharvesting and habitat degradation (Delvaux and Sinsin, 2009; Hamilton, 2004; IUCN, 2014; Street and Prinsloo, 2013; Sunderland and Ndoye, 2004). Since many people depend on herbal medicine for their health and income, the sustainable

extraction of this commodity is essential. The Secretariat of the Convention of Biological Diversity (2010) and the FAO (2009) have called for direct action to conserve the vulnerable and culturally-valued species and their habitats to safeguard key provisioning ecosystem services, particularly those of importance to low income populations.

For most medicinal plant species, however, data on the effects of commercial extraction on their natural populations are unavailable. Even for the most commonly traded species of West Africa, detailed distribution maps do not exist. In order to guarantee a continuous supply of herbal medicine in the future, appropriate management plans must be designed, for which specified information on species occurrence and extraction localities is needed. Understanding trade networks is another key element in designing practical conservation and resource management plans for commercial species (Cunningham, 2001). Quantitative surveys that address volumes of medicinal plants traded have only been carried out for a few African countries, such as South Africa (Williams

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et al., 2007), Tanzania (McMillen, 2008), Ghana (Adu-Tutu et al., 1979; van Andel et al., 2012), Benin (Quiroz et al., 2014), Cameroon (Ingram and Schure, 2010; Ingram et al., 2012a), Gabon (Townsend et al., 2014a) and Central Africa (Clark and Sunderland, 2004; Ingram et al., 2012b). Apart from heavily forested Gabon, where most vendors harvest their own stock (Townsend et al., 2014a), source areas for other African countries are largely unknown. As market chains tend to be long and complex, most vendors purchase their commodities from intermediaries and remain unaware of exact harvesting locations (Belcher and Schreckenberg, 2007; Vodouhê et al., 2008; van Andel et al., 2012; Ingram et al., 2012b; Quiroz et al., 2014).

In the absence of detailed occurrence data, species distribution models (SDMs) are a promising tool to estimate the ecological niche of a species and to predict its reciprocal geographical distribution (Elith and Leathwick, 2009a,b; Franklin, 2009; Guisan et al., 2013). The maximum entropy algorithm is widely used by ecologists and conservation biologists to analyze species' niches and to map their geographic distribution using estimated habitat suitability values (Phillips et al., 2006). It uses presence only data and performs well when few presence records are available (Aguirre-Gutiérrez et al., 2013; Elith et al., 2011; Wisz et al., 2008). Recently, SDMs have been used to map the potential ecological niches of endangered medicinal plant species (Babar et al., 2012; Ray et al., 2011), to predict the future availability of non-timber forest products in relation to climate and land use change (Heubes et al., 2012), and to identify areas of specific cultural value to Australian Aborigines related to the abundant occurrence of medicinal plants (Gaikwad et al., 2011). Herbarium vouchers with detailed collection localities are of great importance to perform these analyses.

The aim of this study was to use quantitative surveys of herbal markets in Ghana and Benin and SDMs to prioritize West African medicinal plant species for field studies on sustainability and conservation needs. More specifically, we sought to answer the following questions: (1) What are the distributions of commercially valuable medicinal plants in Ghana and Benin? (2) Can predicted distributions of these plants be used to assess their vulnerability to overharvesting? (3) Are spatial analyses useful to prioritize areas to study the sustainability of plant extraction in West Africa? (4) Can we suggest an IUCN threat status for heavily exploited medicinal species based on our results?

Our study contributes to the selection of priority species for sustainable management and the identification of probable extraction areas that could serve as a starting point for ground truthing of our models. Our models facilitate the mapping of provisioning ecosystem services, essential for the design of adaptive management strategies in regions where the extraction of non-timber forest products (such as medicinal plants) is crucial to local people's basic needs (Secretariat of the Convention on Biological Diversity, 2010; Guisan et al., 2013).

2. Material and methods

2.1. Species selection and occurrence data

A list of 14 medicinal species was selected that were in high commercial demand (reflected by the highest volumes traded on domestic markets in Ghana and Benin), harvested from the wild from unknown provenances, and naturally occurring in relatively undisturbed natural vegetation types, such as forest and savanna (Table 1). The combined criteria 'high economic trade value' and '(potential) unsustainable harvesting' were also used by Ingram et al. (2012a) to define 'priority non-timber forest products'. Data on daily volumes offered for sale were retrieved from quantitative inventories of 49 market stalls: 27 in Ghana, surveyed in 2010 (van

Andel et al., 2012) and 22 stalls in Benin, surveyed in 2011 (Quiroz et al., 2014). Information on threat status of the selected species was retrieved from the IUCN Red List of Threatened Species (2014), and CITES (2014). Interviews with herbal medicine vendors and consumers provided additional information on (perceived) scarcity of species in Ghana and Benin (Quiroz and Van Andel, 2014; Quiroz et al., 2014; van Andel et al., 2012). Occurrence data for the 14 selected species (accepted names and synonyms) with their geographical coordinates were mined from the Global Biodiversity Information Facility Portal (GBIF, 2014) using the *gbif* function of the *dismo* library (Hijmans et al., 2005) in R (R Development Core Team, 2012). Our GBIF extract includes all collections of the 14 species from the BRAHMS database of Naturalis Biodiversity Center, as well as other major herbarium databases (e.g., Tropicos). We added a few localities of recent collections made by the authors but not yet uploaded in BRAHMS and GBIF. All collection localities are shown in Fig. 2.

2.2. Environmental predictors

To avoid modelling truncated niche dimensions as a result of delimitating the study area by 'artificial' political boundaries (Raes, 2012), we set the study area from West to Central Africa, ranging from 15° North to 10° South and from 17° West to 35° East, covering almost the entire species' ranges (Fig. 1). We used two datasets to extract the environmental predictors that were used in the SDMs: the 19 bioclimatic variables of the WorldClim dataset (www.worldclim.org; Hijmans et al., 2005) and the ISRIC Soil Database (<http://www.isric.org/>; Nachtergaele et al., 2009), both at a spatial resolution of 5 arcmin ($\sim 9.3 \times 9.3$ km at the equator). To account for possible collection effort biases we used the target background sample approach (Phillips et al., 2009). For that purpose we extracted all collection localities in the study area from collections in the BRAHMS database of Naturalis Biodiversity Center, resulting in 37,650 unique collection localities at the 5 arcmin spatial resolution. This layer indicates the presence of botanical collection localities and was added to the environmental predictors as a mask layer excluding all raster cells without botanical collections. To prevent problems with multi-collinearity that can result in model over-fitting (Dormann et al., 2013), we retained a set of 19 environmental predictors that were uncorrelated at collection localities ($|Spearman's r| \leq 0.7$; Table S1). The selected WorldClim variables were: (1) Mean diurnal range (Bio2), (2) maximum temperature of warmest month (Bio5), (3) minimum temperature of the coldest month (Bio6), (4) mean temperature of wettest quarter (Bio8), (5) annual precipitation (Bio12), (6) precipitation of driest quarter (Bio17), (7) precipitation of warmest quarter (Bio18), and (8) precipitation coldest quarter (Bio19). The selected ISRIC variables were: (9) Exchangeable Aluminium percentage – % of ECEC (ALSA), (10) bulk density (BULK), (11) coarse fragments % >2 mm (CFRAG), (12) C/N ratio (CNrt), (13) Effective CEC (ECEC), (14) electrical conductivity (ELCO), (15) exchangeable Na percentage – as % of CECs (ESP), (16) pH in water (PHAQ), (17) sand % (SDTO), (18) available water capacity (TAWC), and (19) organic carbon content (TOTC).

2.3. Species distribution models

To assess the conservation priorities for medicinal plants in West Africa we used species distribution models (SDMs). A SDM identifies the relationship between (1) species' presence records and (2) environmental conditions at the collection sites of that species. The projection of the identified relationships in geographic space allows predicting where habitat conditions are suitable for a species to occur. From the suite of possible modelling algorithms we selected MAXEnt version 3.3.3k (Elith et al., 2011; Phillips et al.,

Table 1

The 14 commercially valuable species initially selected for species distribution modelling and their characteristics. Traded quantities in kg.

Species	Family	Growth form	Part sold	Use	Habitat	Timber use	Locally perceived as scarce	Volumes offered for sale daily (kg)						
								Ghana	Benin	Threat status IUCN/CITES	Sample size	null AUC	Sign.	% of natural vegetation remaining
<i>Acridocarpus smeathmannii</i> (DC.) Guill. & Perr.	Malpighiaceae	Liana	Root	Aphrodisiac	Forest, savanna		Benin	0	793	Not accessed	77	0.670016	1	78
<i>Daniellia ogea</i> (Harms) Holland	Leguminosae	Tree	Bark, resin	Ritual	Forest	x		565	0	Not accessed	6	0.902933	0	73
<i>Daniellia oliveri</i> (Rolfe) Hutch. & Dalziel	Leguminosae	Tree	Bark, resin	Ritual	Savanna	x	Benin	124	339	Not accessed	75	0.683321	1	65
<i>Entada gigas</i> L.	Leguminosae	Liana	Seeds	Ritual	Forest			89	496	Not accessed	55	0.709329	1	81
<i>Khaya senegalensis</i> (Desv.) A. Juss.	Meliaceae	Tree	Bark, wood	Aphrodisiac, fever, ritual	Savanna, gallery forest	x	Benin	788	12	IUCN Vulnerable A1cd	62	0.691084	1	66
<i>Mondia whitei</i> (Hook.f.) Skeels	Apocynaceae	Liana	Root	Aphrodisiac, ritual	Forest		Ghana	23	93	Not accessed	47	0.710245	1	79
<i>Okoubaka aubrevillei</i> Pellegr. & Normand	Santalaceae	Tree	Bark, seeds	Ritual, aphrodisiac	Forest	x	Ghana	241	0	Not accessed	21	0.782393	1	73
<i>Pericopsis elata</i> (Harms) Meeuwen	Leguminosae	Tree	Leaves, wood	Ritual	Forest	x	Ghana	638	0	Endangered A1cd; CITES Appendix II	13	0.835031	0	81
<i>Piper guineense</i> Schumach. & Thonn.	Piperaceae	Liana	Wood, seeds	Spice, asthma, convulsions	Forest			625	63	Not accessed	160	0.625191	1	79
<i>Pteleopsis suberosa</i> Engl. & Diels	Combretaceae	Tree	Bark	Menstrual pain, stds	Savanna		Benin	719	136	Not accessed	60	0.688315	1	58
<i>Pterocarpus erinaceus</i> Poir.	Leguminosae	Tree	Bark	Anemia, cough, women's health	Savanna	x	Benin	0	1097	Not accessed	103	0.658376	1	64
<i>Securidaca longipedunculata</i> Fresen.	Polygalaceae	Tree	Root	Aphrodisiac, ritual	Savanna		Ghana	561	34	Not accessed	118	0.648898	1	70
<i>Sphenocentrum jollyanum</i> Pierre	Menispermaceae	Shrub	Root	Aphrodisiac	Forest		Ghana	126	36	Not accessed	36	0.755858	1	43
<i>Vitellaria paradoxa</i> C.F. Gaertn.	Sapotaceae	Tree	Bark, seeds, fat	Cough, anemia, skin cosmetic	Savanna			450	20	IUCN Vulnerable A1cd	101	0.657899	1	61

Fig. 1. The study area covering the ranges of the 14 selected commercially exploited species. Landcover categories from the European Space Agency GlobCover Portal (<http://due.esrin.esa.int/globcover/>): 11: Post-flooding or irrigated croplands (or aquatic); 14: Rainfed croplands; 20: Mosaic cropland (50–70%)/vegetation (grassland/shrubland/forest) (20–50%); 30: Mosaic vegetation (grassland/shrubland/forest) (50–70%)/cropland (20–50%); 40: Closed to open (>15%) broadleaved evergreen or semi-deciduous forest (>5 m); 50: Closed (>40%) broadleaved deciduous forest (>5 m); 70: Closed (>40%) needleleaved evergreen forest (>5 m); 90: Open (15–40%) needleleaved deciduous or evergreen forest (>5 m); 100: Closed to open (>15%) mixed broadleaved and needleleaved forest (>5 m); 110: Mosaic forest or shrubland (50–70%)/grassland (20–50%); Mosaic grassland (50–70%)/forest or shrubland (20–50%); 130: Closed to open (>15%) (broadleaved or needleleaved, evergreen or deciduous) shrubland (<5 m); 140: Closed to open (>15%) herbaceous vegetation (grassland, savannas or lichens/mosses); 150: Sparse (<15%) vegetation; 160: Closed to open (>15%) broadleaved forest regularly flooded (semi-permanently or temporarily) – Fresh or brackish water; 170: Closed (>40%) broadleaved forest or shrubland permanently flooded – Saline or brackish water; 180: Closed to open (>15%) grassland or woody vegetation on regularly flooded or waterlogged soil – Fresh, brackish or saline water; 190: Artificial surfaces and associated areas (urban areas >50%); 200: Bare areas; 210: Water bodies; 220: Permanent snow and ice; 230: No data (burnt areas, clouds).

2006) because it was developed to make use of presence-only data, and it is robust against (some) georeferencing errors (Graham et al., 2008). Moreover, it has shown to outperform most other modelling algorithms (Aguirre-Gutiérrez et al., 2013; Elith et al., 2011, 2006), even when few collection records are available (Wisze et al., 2008). We modified the MaxEnt default settings as follows: We only used linear and quadratic features (following recommendations in Merow et al. (2013) and we used all records for model training, i.e. no data partitioning. As a measure of model performance, we used the area under the receiver operating characteristic curve (AUC value), a widely used and unbiased summary metric of model performance for binary data. AUC values run from zero to one when applied to presence/absence data, where value 0.5 indicates no better than random prediction, and value 1 perfect model fit. However, the use of AUC values is unreliable when applied to presence-only data, as is the case here (Lobo et al., 2008; Phillips et al., 2009; Raes and Steege, 2007). Therefore we tested all SDMs for significant deviation from random expectation using a bias corrected null-model (Raes and Steege, 2007). Significant SDMs were projected to the full environmental dataset to predict their probability of occurrence throughout the study area; also for areas where no botanical collections have been made. Finally, the projected SDMs were converted from continuous MaxEnt habitat suitability predictions to discrete presence/absence values, using the 10 percentile training threshold. This conservative threshold forces 10% of the collection presence records outside the predicted presence area, accounting for errors in taxonomic identifications and in georeferencing of the collection localities.

2.4. Setting conservation priorities

To assess what portion of the species' potential distributions was still covered by natural vegetation, we downloaded the Global Land Cover 2009 V2.3 data (<http://due.esrin.esa.int/globcover/>). This dataset recognizes 23 different vegetation classes (Fig. 1). We excluded 'urban areas' and all '>50% cropland' categories on

grounds of being non-natural vegetation cover; the remaining categories represented natural vegetation cover. We projected the natural vegetation cover layer on all significant SDMs, and calculated the percentage of their potential ranges still covered by natural vegetation. This procedure indicated the extent to which a species' natural range had disappeared and therefore its vulnerability to overharvesting. For that purpose we used the thresholded potential MaxEnt prediction as original 'Area of Occupancy' (AOO) (Syfert et al., 2014) and we assessed how much of each species' original range has been lost to land-use change. It should be kept in mind that range reduction is not necessarily proportional to population reduction. Locally population densities can be low due to overharvesting resulting in a higher IUCN threat status level than would be expected based on SDM range estimate reduction as a results of land use change. Therefore, species are likely more threatened in reality. For species that were not previously assessed by the IUCN, we suggested IUCN threat status levels based on natural habitat loss within their estimated ranges of occurrence and the criteria provided by the IUCN Red List Categories and Criteria (IUCN, 2012). This procedure allows the use SDMs range estimates directly to guide conservation strategies, as suggested by Guisan et al. (2013).

3. Results

3.1. SDMs of the selected species

In total, the 14 selected species were represented by 934 unique occurrence records, ranging from six to 160 occurrences per species. From our initial list of 14 species, 12 had AUC values that were significantly higher than random expectation ($p \leq 0.05$; Table 1). Tree species *Daniellia ogea* and *Pericopsis elata* (listed on CITES Appendix II and considered Endangered A1cd by the IUCN) were excluded from further analysis because their AUC values did not surpass the significance threshold of the bias corrected null-model. The 12 remaining commercial species

Fig. 2. MaxEnt presence predictions of the 12 commercially harvested species in West Africa with a significant SDM. Dots indicate the collection localities, hatching indicates natural vegetation cover and non-hatching indicates cropland.

(four lianas, one shrub and seven trees) represented an average volume of 349 kg (Ghana) and 262 kg (Bénin) of bark, roots, wood, resin and/or seeds offered for sale on a daily basis at domestic markets. Of the 12 selected species, five occur in forests and five in savannah, while two can be found in both habitat types. Six of the seven tree species were also extracted for timber (ref?).

The distribution maps generated by MaxEnt (Fig. 2) show that six of the commercial species (*Daniellia oliveri*, *Khaya senegalensis*, *Pteleopsis suberosa*, *Pterocarpus erinaceus*, *Securidaca longipedunculata* and *Vitellaria paradoxa*) occur in a wide range of suitable habitats. These habitats correspond with the West African savanna belt ranging from the northern (drier) parts of Ivory Coast and Ghana to the Dahomey gap and cover all but the wettest parts of

southern Benin. Apart from the lianas *Mondia whitei* and *Acridocarpus smeathmanii*, which seem to occur throughout West Africa except for the driest zones, the remaining four species (*Piper guineense*, *Sphenocentrum jollyanum*, *Entada gigas* and *Okoubaka aubrevillei*) were confined to the evergreen upper Guinean forest, stretching from southwestern Ghana to Sierra Leone and the Central African rainforests from Cameroon to the Congo Basin (Fig. 2).

3.2. Savanna species and vulnerability to overharvesting

In spite of the wide potential distribution of *S. longipedunculata* (Fig. 2) predicted by our SDMs, market vendors in Ghana reported that its roots, highly valued as an ingredient for aphrodisiac mixtures (van Andel et al., 2012), were becoming rare due to overexploitation. The species is however not listed by IUCN as threatened (Table 1). The bark of *K. senegalensis* was previously collected in the wild, but due to scarcity it is now increasingly harvested from individuals planted as shade trees along the major roads in Accra and Cotonou. The tree is listed as Vulnerable A1cd by IUCN. *V. paradoxa* is commonly grown in savanna parklands as a source for shea butter in northern and central Ghana and Bénin (Schreckenberger, 2004), but wild stocks have been overexploited for timber, firewood and charcoal production (IUCN, 2014). Its habitat is also suffering from agricultural encroachment and increasing population pressure (<http://www.iucnredlist.org/details/37083/0>). The recent land cover data used in our analysis reveal that for the savanna species in this study, 21–39% of their range is no longer covered with natural vegetation. This provides support for local people's perception of scarcity, even for species with wide predicted distributions. Several of our selected species with wide predicted distributions were given a status of 'vulnerable' (*M. whitei*) or even 'endangered' (*A. smeathmanii*, *K. senegalensis*, *P. erinaceus*) on the National Red List of Bénin, because of their extensive commercial extraction and habitat deterioration (Adomou et al., 2011).

3.3. Forest species and vulnerability to overharvesting

Although the Brahms database contained two records of *P. guineense* (Fig. 2) for Benin, this was not reflected in predicted presence. The spicy grains of this forest liana, sold frequently at Beninese urban markets as a condiment and ingredient for medicinal mixtures, were probably imported from Cameroon, Ghana or Ivory Coast, where it occurs more widely. In Cameroon, this species was considered a priority NTFP, a wild-harvested product from lowland forest with substantial economic value (Ingram et al., 2012a). Although small-scale cultivation of *P. guineense* is practiced by local farmers, the bulk of the harvested material comes from wild resources (Ingram and Schure, 2010). Although Central African harvesters typically uproot the plant and strip all its seeds, destroying the plant and reducing regeneration (Clark and Sunderland, 2004), the harvest of *P. guineense* was still considered to have a relatively low impact on natural populations (Ingram and Schure, 2010).

E. gigas (Fig. 2), also known as sea heart, is a robust woody climber, occurring in forests on sandy soils along rivers and sea coasts. Its large pods and seeds are dispersed by ocean currents. The species is also found along beaches in Central America and the Caribbean (<http://www.tropicos.org/Name/13009179>). Given its limited distribution in Benin and the high demand of *Entada* seeds on the market for ritual purposes, it is likely that they are imported from Cameroon, Ghana, Central Africa or Liberia, where the liana still has a substantial predicted range (Fig. 2). Neither of the two lianas (*P. guineense* and *E. gigas*) have been assessed by IUCN. Although 80% of the natural vegetation of these two species is still intact (Table 1), West African coastal forests are rapidly cleared for agriculture and urban expansion, visible as the

non-hatched areas of cropland in Fig. 2. For this reason and their commercial value, we suggest for both species an IUCN status of Vulnerable A1cd for the West African populations. Populations of the two lianas in Central Africa are probably less threatened, but they appear to be isolated from the West African ones by the Dahomey gap (Fig. 2).

S. jollyanum (Fig. 2) has the narrowest distribution of the selected species, occurring only in small patches of remaining forests in the southern parts of Ivory Coast, Ghana, Togo, Benin, and Nigeria. It has a limited predicted presence in Cameroon, and is poorly documented with herbarium collections. Due to its occurrence in the shaded understory of rainforests, its small range, the commercial extraction of its roots as aphrodisiac, and the fact that only 43% (Table 1) of the natural vegetation remains in its predicted distribution, we suggest an IUCN status of Endangered A3cd for *S. jollyanum*. This status is defined as facing a high risk of extinction in the wild, because of an estimated populations size reduction towards $\geq 70\%$, an ongoing decline in the quality of habitat and actual levels of exploitation (IUCN, 2012). Currently the species has not yet been assessed by the IUCN. The species was mentioned earlier as locally threatened by Schmelzer and Gurib-Fakim (2008).

The forest giant *O. aubrevillei* (Fig. 2) is worshipped as a sacred tree by local people in southern Ghana. Its seeds and bark are extracted for medicinal and ritual purposes, whereas the species is being logged illegally for timber (Myren, 2011). Its natural habitat, primary rainforest from eastern Liberia to southwest Ghana and the Congo Basin, is still 73% intact (Table 1). *O. aubrevillei* has not yet been assessed by the IUCN. However, it is rare throughout its range, it has poor natural regeneration, international demand for its timber and medicinal bark is high, and its natural habitat faces continuing deterioration (Schmelzer and Gurib-Fakim, 2008). Therefore, we suggest a new IUCN status of Vulnerable A3cd for *O. aubrevillei*, confirming the earlier warnings on the vulnerability of this species (Schmelzer and Gurib-Fakim, 2008).

Based on their commercial value and their restricted distributions, *S. jollyanum*, *O. aubrevillei*, and to a lesser extent *E. gigas*, *K. senegalensis* and *P. guineense*, emerge as priority species for conservation. *S. jollyanum* has the smallest range of all species, is killed during harvest as people pull out seedlings with their entire root, and 57% of its natural habitat has been destroyed. The forests where *O. aubrevillei* and *E. gigas* occur are more than 70% intact. Populations of these three species that are located close to expanding urban areas (large markets) or roads (easy transport) are the most vulnerable for overharvesting.

4. Discussion

4.1. Using SDMs to predict species' vulnerability and NTFP availability

Our study provides the first detailed predicted occurrence maps of major commercially traded medicinal plant species in West Africa. Facing the present rates of deforestation and land use change in West Africa (Grainger, 2013), the patchy occurrence of some of these species (*S. jollyanum*, *O. aubrevillei* and to a lesser extent *E. gigas* and *P. guineense*) may further increase their economic value. This will probably increase harvesting intensity and subsequent vulnerability, which justifies a close monitoring of their exploitation.

Although plant occurrence predictions by SDMs do not provide information on species abundance within their suitable habitats, preliminary quantitative analyses indicate a possible relationship between habitat suitability and abundance (Martínez-Meyer et al., 2013). Tree species with (predicted) broad ecological niches like *K. senegalensis*, *P. erinaceus* and *S. longipedunculata* are not

indicated as species of concern by these models. Nonetheless, commercial exploitation for timber and medicinal products can greatly affect populations of these trees (Louppe et al., 2008). Bark harvesting for medicinal purposes often follows logging operations, in which the bark can be considered a by-product of destructive timber extraction, like in the case of *Pausinystalia johimbe* in Central Africa (Sunderland et al., 1997). Bark removal alone, however, can also seriously affect tree populations. In Benin, *K. senegalensis* individuals with a diameter at breast height (dbh) > 30 cm were said to be scarce, even in protected areas (Delvaux et al., 2010). This is a remarkable contrast, as saplings of this species are considered as locally abundant with ca. 43 individuals with a dbh > 5 cm per hectare (Gaoue and Ticktin, 2007a). This suggests that although young individuals are abundant, adults are intensively harvested for their bark, wood and leaves. For *P. erinaceus*, detailed population figures are lacking for our study area, but research in Senegal showed that regeneration is severely affected in areas with frequent fires and strong human impact (Lykke, 1998). *S. longipedunculata*, the savanna tree with the widest distribution of our selected species, apparently does not respond well to the regular removal of roots. Seeds germinate with difficulty while seedlings grow slowly and do not tolerate transplanting, which makes the species difficult to cultivate (Zulu et al., 2011).

The African Sahel zone is expected to face severe changes in climatic conditions and land use this century, which will strongly affect provisioning ecosystem services and availability of NTFPs (Heubes et al., 2012). Savanna trees that are currently abundant and widespread, such as *V. paradoxa*, may still face substantial niche reduction in the future. Heubes et al. (2012) predicted that climate and land use changes, such as decreasing water availability, raising temperatures and changes in land tenure, will strongly influence the availability of NTFPs from savanna trees in northern Bénin. The marketing of shea butter from *V. paradoxa* seeds as a locally sourced cooking fat or for the international cosmetic industry now represents up to 13% of the annual income of women in this region, but monetary losses of 50% were predicted for *V. paradoxa* by 2050, with immediate consequences for the income generation of local harvesters (Heubes et al., 2012). According to Schreckenberg (2004), however, it is the market that determines the future of *V. paradoxa*, as most of the shea butter comes from trees that are planted or actively managed. Niche reduction due to climate change may affect some of the commercial savanna species treated in our survey (e.g., *P. suberosa*, *Daniellia oliveri*) which are presently still considered abundant (Bognounou et al., 2009; Lykke, 1998; Laird et al., 2010). SDMs that are projected to predict future climate and land use conditions can be of great value to monitor and manage the future availability of NTFPs, but when (semi-) cultivated plants are concerned, the market for their products and their agricultural growing conditions should be taken into account as well.

4.2. SDMs as a tool to prioritize areas to study sustainable extraction activities

As transport costs are a major constraint in successful NTFP commercialization (Belcher and Schreckenberg, 2007; van Andel, 2000), medicinal plants are preferably harvested as close as possible to urban areas and roads, from where they are transported to markets (Shanley et al., 2002; van Andel et al., 2007). In sparsely populated areas like Suriname or Gabon, where savannas or forests are located close to cities, market vendors harvest their own herbal medicine, which makes it relatively easy for researchers to locate extraction areas and study harvesting methods (Townsend et al., 2014a; van Andel and Havinga, 2008). In densely populated and largely deforested areas, such as the West African coast, market vendors may harvest weeds and cultivated plants themselves,

but depend on intermediaries for their supply of forest and savanna species (Quiroz et al., 2014; Townsend et al., 2014b; van Andel et al., 2012). Here, market chains tend to be long and complex, and extraction sites for commercial species are difficult to locate.

Our models have shown that SDMs can be helpful to indicate priority areas to study the sustainability of plant extraction from the wild, especially for species with restricted distributions (Sarkinen et al., 2013). Field studies on the sustainability of the extraction of *E. gigas* seeds, *P. guineense* seeds, *S. jollyanum* roots and *O. aubrevillei* seeds and bark should be directed towards patches of remaining coastal forests located near large cities, such as Accra, Cotonou, Port Novo, Lomé and Lagos, or along main roads leading towards these urban centres. Even for widely distributed species, it is likely that most extraction takes place near urban centres and roads to reduce transport costs. Existing trade channels, the presence of people willing to harvest, accessibility and local land tenure systems shape extraction patterns for NTFPs (Vodouhê et al., 2008). Combined with market chain analysis (such as the detailed studies on *Gnetum* leaves in Cameroon by Ingram et al. (2012b) and shea butter by Schreckenberg (2004)), seasonal availability and land tenure studies, our models can be used as a basis to map extraction probability and trade routes of commercially valuable species in further detail. Local initiatives for cultivation of NTFPs (like in the case of *V. paradoxa* and *P. guineense*) should also be taken into account, as these may result in the occurrence of species outside their predicted niche.

4.3. Monitoring extraction activities based on SDMs

Although SDMs play a key role in identifying areas with suitable abiotic conditions, it remains essential to realize that SDMs do not replace ground truthing. Additionally, a species can be absent from areas with suitable abiotic conditions due to lacking biotic interactions (i.e. pollinator is missing), competitive exclusion by other species, inaccessibility due to geographical or (a)biotic barriers, or overharvesting by humans (Araújo and Peterson, 2012). To find out whether the commercial collection of wood, roots or seeds is a destructive activity, field studies on species' abundances, and the impact of different extraction methods on the survival, growth and reproduction of the harvested individuals remain essential (Delvaux et al., 2010; Gaoue and Ticktin, 2007b).

Bark harvesting, for example, may be sustainable as long as trees are not ring-barked or felled and species are able to recover quickly. Previously untouched individuals of *K. senegalensis* in Bénin showed very good bark recovery rates and complete wound closing after 24 months (Delvaux and Sinsin, 2009). Because of its resilience to debarking and its fast growth in open areas, *K. senegalensis* is said to have the potential to support sustainable bark harvesting (Delvaux et al., 2010). However, this tree is frequently harvested simultaneously for its bark and leaves by local herders to feed their livestock, which negatively affects seedling and sapling densities (Gaoue and Ticktin, 2007a). This situation suggests that continued harvesting in some populations may present conservation concerns over the long-term. *P. erinaceus* showed a slower response to bark removal, but since the species is able to sprout back after felling, Delvaux et al. (2010) suggested cutting trees at 1 m height and stripping off their bark, after which they would coppice from the trunks and generate new individuals over time. For the other 10 commercial species, information on the ecological impact of bark and leaf harvesting or suitability for cultivation is still lacking.

In the current study, each species was considered independently from the specific purpose for which the species was being harvested and the presence of alternative products. In some cases there may be substitute plant products to replace the focal species

as sources for the desirable medicines. However, the degree by which substitutes are available (and acceptable for the local consumers) is currently unknown and deserves further research.

5. Conclusions

Our species distribution models provide the first high spatial resolution distribution maps for commercially valuable, wild-harvested medicinal plants in West Africa. When combined with quantitative market data, SDMs are useful tools to identify species that are vulnerable to overharvesting because of limited range distributions. Two narrowly-distributed species restricted to closed evergreen forests, *S. jollyanum* and *O. aubrevillei*, emerged from our models as priority species for conservation and sustainable extraction. We suggest a new status of Endangered A3cd for *S. jollyanum*, and Vulnerable A3cd for *O. aubrevillei*. For *E. gigas* and *P. guineense*, two lianas with a high market demand and limited distribution in West Africa, but a much larger predicted area in the Congo Basin, we suggest an IUCN status of Vulnerable A1cd. None of these species are currently listed by CITES or the IUCN.

As SDMs emphasize extent of suitable areas, not abundance, they are of limited use to assess vulnerability for overharvesting of widely distributed species. Heavily exploited savanna trees (e.g., *K. senegalensis* and *P. erinaceus*) did not emerge as species of conservation concern from our models, as their broad environmental suitability misleadingly suggests less anthropogenic threat. Local perceptions of scarcity among market vendors and harvesters may be better indicators to prioritize species for conservation concern in such cases.

If species vulnerability is expressed as a function of reduced predicted habitat and high commercial demand, SDMs are helpful to select priority areas for ground truthing. Field surveys should start in predicted habitats that are close to urban areas and main roads, as commercial extraction is most likely to occur at the shortest cost distance to the market. However, SDMs alone can never indicate extraction localities with 100% certainty, as these are also determined by existing trade channels and land tenure. Even when species have restricted habitats, this does not mean that their harvest is always destructive. Field studies are needed to verify predictions, measure abundance and determine if current harvesting practices negatively affect population size and jeopardize their sustainable use.

Our study shows that SDMs for commercially harvested medicinal plants are a useful tool to select priority species for conservation and sustainable extraction, but also serve to identify the geographical locations of provisioning ecosystem services that are vital to local people's livelihood. Such models enable a visualization of the *Economics of Ecosystems and Biodiversity* (TEEB, <http://www.teebweb.org/>), which can help policymakers to safeguard these valuable species and their habitats in the future by proper land use management.

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Our funding source (NWO) had no involvement in the design of our study, the collection, analysis, and interpretation of data, in the writing of the report or in the decision to submit this manuscript for publication.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.biocon.2014.11.015>.

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