Title: Assessing the extent of access and benefit sharing in the wildlife trade: lessons from horticultural orchids in Southeast Asia

Short title: Access and Benefit Sharing in the Orchid Trade

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Word count (including references and legends): 5,991

Summary

The equitable sharing of benefits from natural resources is a key target of the Convention on Biological Diversity. Trade in its native species is one way in which a country can potentially benefit from its natural resources, and even small-scale traders can now access global markets online. However, little is known about the extent of benefit sharing for many products, and to what extent the appropriate processes and permits are being used. We surveyed online trade in a lucrative and widely-sold product in Southeast Asia (horticultural orchids), to assess the extent of access and benefit sharing. In total, 20.8% (n=1120) of orchid species from the region were being sold. Although 7/10 countries were trading, five had very little or no trade in their native species, and the majority of recently described endemic species being traded from non-range states had no reported CITES exports from their country of origin. We suggest that addressing access and benefit sharing gaps requires wider recognition of the problem, coupled with capacity building in the countries currently benefitting least: Laos, Myanmar and Cambodia. The priority should be to increase botanical capacity and enable these countries to better control the commercialization and trade of their native species.

INTRODUCTION

Commercial trade of its native plant and animal species is one way in which a country can gain economic benefits from its natural genetic resources. These benefits may include direct income to companies and individuals participating in trade, but also wider benefits such as increased income from taxes (Laird & Lisinge, 1998), greater in-country business spending (e.g. on rent or materials), and creation of jobs in supporting industries (Jepson et al., 2011). The sovereign right of a country to sustainably exploit its natural genetic resources, and benefit when these resources are used by others, is known as Access and Benefit Sharing (ABS), and is one of the three core objectives of the Convention on Biological Diversity (CBD) (CBD, 1992; Nagoya Protocol, 2011). Exploitation of another country's natural resources usually involves collection of wild material to supply companies directly, or to enable artificial propagation or captive breeding *ex situ* (Laird & Lisinge, 1998; Trommetter, 2005). Where the use of these resources takes place formally, ABS principles require compensation, which may include up-front or ongoing payments, royalties from sales (Trommetter, 2005), or the transfer of knowledge, goods or technology to build capacity for trade within the country of origin (FAO, 2009). However, shifts in trade networks, product types, and methods of trade have taken place since the CBD came into force over two decades ago, some of which are likely to add further complexity to ABS implementation. A good example is the rapid increase in online wildlife trade, a development that has provided opportunities for small businesses to access international markets, but which has proved difficult to monitor and regulate (Lavorgna, 2014).

58 Identifying and addressing ABS inequities is important, not only because benefit
59 sharing is an ethical issue (Schroeder, 2007) but also because in some cases it has
60 the potential for tangible conservation benefits, for example by providing an incentive
61 for the protection of exploited species and habitats (e.g. butterflies: Gordon &
62 Ayiemba, 2003). However, despite its recognised importance, to date there have
63 been few studies of how ABS has worked in real markets. These studies include
64 ABS examples in the agricultural (Richerzhagen & Holm-Mueller, 2005), cosmetic
65 (Lybbert et al., 2002), pharmaceutical and phytomedical (Laird & Lisinge, 1998), and
66 food-supplement sectors (Vermeylen, 2007). However, efforts to assess the extent
67 and form of ABS in other markets that rely on the development of new products from
68 wild genetic resources are limited. One such market is the international horticultural
69 trade, which has a relatively limited awareness of ABS (Ten Kate & Laird, 2000;
70 Secretariat of the Convention on Biological Diversity, 2008), despite clear emphasis
71 on the importance of benefit sharing by the Global Strategy for Plant Conservation
72 (CBD, 2002; CBD, 2012). The horticultural trade is extremely lucrative, with an
73 estimated global export value of US$9.1 billion live plants in 2013 (ITC, 2014).
74 Although most traded plants are mass-produced hybrids, wild species are important
75 in the development of new products, a trend that is predicted to increase as breeding
76 technology improves (Volk & Richards, 2011). The only high profile horticultural ABS
77 case was in 1999 between the South African National Biodiversity Institute (SANBI)
78 and the American company Ball, to jointly develop new products from South Africa's
79 wild flora (Henne & Fakir, 1999).
80
81 Here we focus on ABS in the Southeast Asian orchid market, by studying the online
82 market for orchid species. Orchids are one of the top horticultural plants in trade in

terms of sales volume, net profits and price consistency over time (FloraHolland, 2013; USDA, 2014) and comprise 70% of all species listed by the Convention on the International Trade in Endangered Species of Wild Fauna and Flora (CITES, 2013). However, even though all orchid species are listed by CITES, their trade receives little attention (Phelps & Webb, 2015). In addition, they are relatively easy to transport across international borders, as they are difficult to identify (McGough et al. 2006) and likely to be a low priority for busy customs officers. Orchids are also widely traded online, including some trade which does not comply with national and international trade regulations (Krigas et al. 2014; Hinsley et al. 2016b). Here we test the use of an online survey to assess ABS for traded products, with the aim of identifying which countries are not trading in their native and endemic species, and which countries are trading in the species of others. We hypothesize that the countries with the least capacity for trade (in terms of paucity of botanical and horticultural expertise, and limited access to propagation technology) will be the ones most likely to be losing out.

METHODS

The internet is increasingly being used to sell plants, animals and other products derived from wildlife (Lavorgna, 2014) including horticultural plants (e.g. Krigas et al. 2014). Trading online allows traders and buyers of illegal products to evade detection (Hinsley et al. 2016b) but online trade also provides a good opportunity for the study of large scale trade patterns. We focus our analysis on Southeast Asia, a hub of legal and illegal wildlife trade (Nijman, 2010), and a centre of diversity for the tropical epiphytic orchid species that are popular in trade, including two species (*Dendrobium cruentum* and *Renanthera imschootiana*) and one genus

(*Paphiopedilum*) listed in CITES Appendix I (CITES, 2013). Studies of orchid trade via street markets in the region have already taken place (e.g. Phelps et al., 2014) but little attention has been paid to the study of internet trade, which is becoming increasingly important for horticultural plants (Sajeva et al., 2013; Shirey & Lamberti, 2011). We focus on 10 Association of Southeast Asian Nations (ASEAN) countries: Brunei Darussalam (hereafter Brunei), Cambodia, Indonesia, Lao Peoples' Democratic Republic (hereafter Laos), Malaysia, Myanmar, the Philippines, Singapore, Thailand and Viet Nam (ASEAN, 2013).

Between April and June 2012 we searched the www.orchidmall.com and www.orchidwire.com vendor directories and carried out Google searches for each country name plus 'orchid nursery', 'orchid for sale' and 'orchid species' (after Shirey & Lamberti, 2011). We then consulted in-country orchid experts to identify any missed nurseries. Due to our focus on ABS, we restricted analysis to trade via official nursery websites, as these are likely to represent formal, although not necessarily legal, trade.

Each website was visited and all orchid species for sale were recorded, including any recognized species listed as parent plants of hybrids. We recorded all species whether wild or artificially propagated, but omitted complex hybrid plants, many of which are mass-produced for non-specialist buyers (Hinsley et al. 2015), and often too far-removed from wild genetic resources for these links to be made. In addition, species are usually aimed at the smaller specialist market, which presents a greater opportunity for small-scale producers. To look at variations in taxonomic accuracy and listing language in each country, we coded each listed name as (1) an accepted

133 species name; (2) a recognized synonym; and (3) an unknown/trade name.
134 Presence/absence and type of descriptors were also recorded, for example whether
135 the listing included a physical description (e.g. flower color/size), geographical
136 (country/region) or other information (e.g. ‘new species’).
137
138 We used the World Checklist of Selected Plant Families (WCSP, 2014) to check
139 taxonomy and species’ distributions, and to compile national lists of native and
140 endemic species. The coding system in this database for distributions matched
141 political boundaries for most countries, with some exceptions. The code for New
142 Guinea did not distinguish between species in Papua New Guinea and Indonesian
143 New Guinea, so all species with this code were omitted unless further detail showed
144 that they were present or endemic in Indonesia. In addition, the Borneo code did not
145 separate Indonesian Borneo, Malaysian Borneo, or Brunei. As these countries were
146 all part of the study this code was included and, where available, additional
147 information in each species listing was used to assign species as present or endemic
148 to one of these countries. For those that could not be assigned, we used a sensitivity
149 analysis to investigate the effect of including these species in either Malaysia or
150 Indonesia. Singapore was listed under the Malaysia code, so Singaporean species
151 were identified using Chong et al. (2009). Finally, East Timor endemics (Silveira et
152 al., 2008) were removed from the Indonesian total.
153
154 We produced descriptive statistics for the region and individual countries, and used a
155 Pearson’s Chi squared Goodness of Fit test to compare each country to the regional
156 figure for the proportion of own native and endemic species that it sold. We then
157 used simple weighted network analysis (Opsahl, 2010) to calculate eight network

158 measures for each country: the out-degree, in-degree, out-strength and in-strength,
159 for both native and endemic species. For each country, the degree is defined as a
160 count of the number of other countries which a) sell that country's native/endemic
161 species (out-degree), and b) have native/endemic species sold by that country (in-
162 degree). Similarly the strength is defined as a count of the number of species a)
163 native/endemic to that country that are sold by other countries (out-strength), and b)
164 sold in that country, which are native/endemic to another country (in-strength).
165
166 Finally, we carried out an analysis of all recently discovered endemic species found
167 for sale outside the country of origin to investigate whether exports have taken place
168 via formal channels, and how rapidly these species are commercialized for
169 international trade by their country of origin. We calculated the time from date of
170 description (WCSP, 2014) to first commercial export reported to CITES from the
171 country of origin (UNEP-WCMC, 2017). Our search was for all exports (importer or
172 exporter reported) of any product that could lead to the production of live plants for
173 trade (live plants, cultures, seeds, roots and stems). We analysed only species
174 described since CITES began in 1975, with a separate analysis of species described
175 since 1996, as better data checks were introduced in late 1995 (UNEP-WCMC,
176 2013). Although Laos only became a party to CITES in 2004, non-Parties are
177 required to have equivalent documents for the export of listed species (Resolution
178 Conf. 9.5 (Rev. CoP16)).
179
180 **RESULTS**
181 We found 87 websites, 49% (n=43) of which were excluded from the analysis
182 because: they only sold complex hybrids or cut flowers (n = 24), were for a related

183 business (e.g. selling pots or fertiliser) (n = 7), were not working for the whole study
184 period (n = 6) or listed no products for sale online (n = 6) (Table 1).
185
186 There were 5387 species reported to be native to at least one country in the region,
187 ranging from 23 in Brunei to 3082 in Indonesia (including all Borneo species) (Fig. 1).

188

189 **Figure 1:** Total number of native orchid species in each country in Southeast Asia
190 (all species listed under the Borneo code with no further information are included in both the
191 Indonesian and Malaysian totals) (data from World Checklist of Selected Plant Families:
192 WCSP, 2013).
193
194 Of this regional total, 20.8% (n = 1120) were found for sale. When Borneo was
195 included in Indonesia, 9.9% of species endemic to at least one country in the region

were in trade; when Borneo was included in Malaysia this figure was 9.6%. The observed proportions of native species sold by country of origin differed significantly from the expected value (Borneo = Indonesia: χ^2 = 979.0, 6 d.f., P<0.001; Borneo = Malaysia: χ^2 = 868.1, 6 d.f., P<0.001). Similarly, sales by each country of their own endemic species differed significantly from the expected, both when the figure used was 9.9% (Borneo = Indonesia: χ^2 = 274.5, 6 d.f., P<0.001; Borneo = Malaysia: χ^2 = 275.8, 6 d.f., P<0.001) and 9.6% (Borneo = Indonesia: χ^2 = 195.0, 6 d.f., P<0.001; Borneo = Malaysia: χ^2 = 195.9, 6 d.f., P<0.001).

Native species from Cambodia, Laos, Myanmar, Indonesia (including Borneo), the Philippines and Malaysia were on sale in every country where trade was occurring (n = 6). Endemic species from Indonesia, Malaysia and the Philippines were on sale in the most countries (5/6 trading countries). Nurseries in Singapore and Malaysia sold native species from every country in the region, whilst Thailand and Singapore sold endemic species from the most other countries (6/9) (Table 2).

We found 137 endemic orchid species for sale in at least one non-range state, of which 21 were described between 1975 and 1995, and 29 between 1996 and 2012. Of the 50 endemic species described since 1975, 32 (64%) had no CITES record of export from their country of origin. For those described after 1996, 21 (72%) had no reported exports form their country of origin (Fig. 2), including 4 CITES Appendix I *Paphiopedilum* species. Most of these 21 species were from Indonesia and Viet Nam. In addition, two species (*Bulbophyllum coweniorum* and *Holcoglossum calcicola*) were from Laos, which had no facilities to produce artificially propagated orchids at this time.

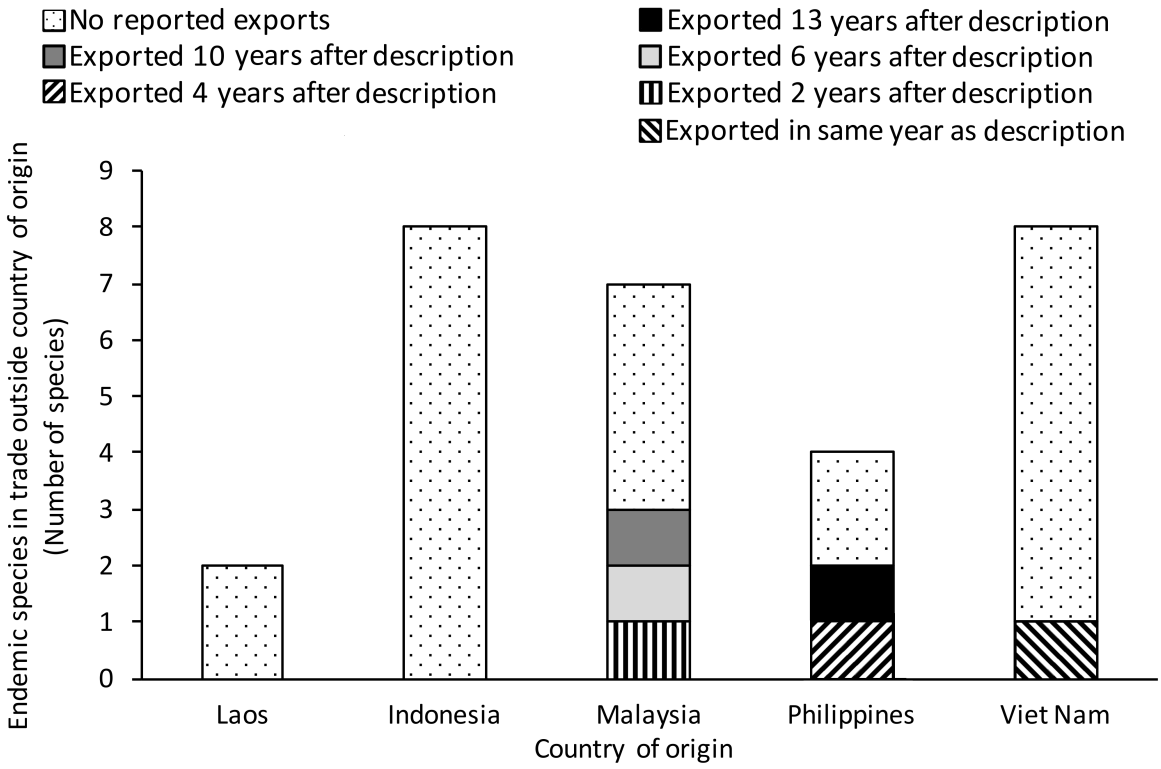

Figure 2: Number of endemic species described 1996-2012 for sale from non-range states, showing a breakdown of the number of years from discovery to first reported CITES export from country of origin (data from CITES Trade Database: UNEP-WCMC, 2017)

DISCUSSION

Our study of the online horticultural orchid trade in Southeast Asia suggests that the international movement and commercialization of species is widespread, with more than 1 in 5 of the region's species found for sale from online platforms. However, much of this trade appears to have taken place without formal ABS implementation and some without CITES permits. This supports earlier concerns of limited awareness of ABS in the horticultural sector (Ten Kate & Laird, 2000; Secretariat of the Convention on Biological Diversity, 2008), and findings of CITES non-compliance in the orchid trade, especially by professional growers (Hinsley et al. 2016b).

Despite the growth of online trade in wildlife products (Lavorgna, 2014), there has been little work to understand how this trade is linked to broader trade patterns. We show that large numbers of species are being sold online and that these numbers are comparable to recorded data on offline trade. For example, Phelps & Webb (2015) found 13% of Thailand's orchid flora for sale during surveys over one year in four large flower markets, compared to our finding of 25.2% of the country's orchid species sold by Thai nurseries online, and 41% sold by nurseries in the whole region. Further, broad patterns observed in online trade can closely match offline data, suggesting that these markets can be a good proxy for understanding broad trade trends in related products. For example, the countries in our study with the most (Thailand, Malaysia and Singapore) and least trade (Myanmar, Cambodia, Lao and Brunei) match customs data for the countries with the highest and lowest value exports of both general horticultural products and cut orchid flowers (ITC, 2014). Despite their utility, we acknowledge that online surveys will not capture the local trends and patterns of trade that can be observed in offline shops and markets (e.g. Phelps et al., 2014). In addition, surveys of online formal online trade may omit important informal platforms where orchid trade takes place, such as social media websites (Hinsley et al, 2016a). However, online sales are playing an increasingly prominent part in horticultural and other wildlife trades (Lavorgna, 2014), and surveying them provides an easily accessible method for the study of these markets (Shirey & Lamberti, 2011; Sajeva et al., 2013; 2013; Krigas et al. 2014). Further work to assess the linkages between online and offline markets for horticultural and other wildlife products is needed to better understand these interaction between.

Our findings suggest that two decades on from the introduction of the CBD, the countries of Southeast Asia are not benefitting equally from trade in their native species. We acknowledge that our focus on formal trade does not recognize the benefits that may be transferred from illegal orchid trade, which may be essential supplementary income for some households (Hinsley, 2011). However, the collection of orchids for trade can quickly become a significant conservation issue without careful management, leading to rapid decline or extinction (Averyanov et al., 2003). These informal agreements may bring short term benefits to some people, but the potential benefits from the commercialization of valuable species will exist over a much longer period, meaning that overall the country is losing out (Laird & Lisinge, 1998). We therefore identify several countries that would benefit from action to address ABS inequities in formal trade, primarily Laos, Cambodia and Myanmar. These findings are likely to be linked to economic development, as Cambodia and Laos have the lowest Gross National Income per capita in the region (no data available for Myanmar) (World Bank, 2014). Identifying the form that ABS activities could take is not straightforward. The Nagoya protocol recommends that equitable sharing of benefits should be achieved by "appropriate transfer of relevant technologies … and by appropriate funding" (Nagoya Protocol, 2011, p4). Other examples for ABS have shown that this often takes the form of direct payments for the bioprospecting of new products (e.g. Richerzhagen & Holm-Mueller, 2005). However, applying the principles of ABS to the orchid trade will require a different approach. For example, direct payments for initial access to, or on-going use of, a country's genetic resources is an approach taken in the pharmaceutical industry (Trommetter, 2005) but has had limited application in the horticultural trade. The landmark agreement between the horticultural company Ball and South Africa's

National Biodiversity Institute (SANBI) eventually resulted in direct benefits being shared, but demonstrated that careful management was essential (Secretariat of the Convention on Biological Diversity, 2008). The company in this case was large and had the resources to make a long-term commitment to fund SANBI. Whilst this may be a useful model for the mass-market horticultural industry, it is unlikely to work for the orchid species market, which is supplied by small businesses selling a large range of species in small numbers. Additionally, direct payments would only be successful for newly commercialized species, as sharing benefits is particularly difficult if captive breeding or propagation has already been taking place for some time in different countries (Roe et al., 2002; Richerzhagen & Holm-Mueller, 2005).

If direct payments are unsuitable, another approach suggested in other ABS cases is capacity building to allow countries to develop their own trade (FAO, 2009). In theory, this approach may address some of the potential causes of the gaps found in our study. For example, we found that the countries with little or no trade in their own taxa contributed a large proportion of their species to the trade of other countries, including over half of Laos' native species and three of its 12 endemic species. This suggests that the gaps in trade are not due to a lack of market for these species but to a lack of interest or capacity for trade. The former is unlikely, as several countries in the region have declared an interest in developing orchid trade (Viet Nam News, 2010; Hajramurni, 2011; The Brunei Times, 2012; Malanes, 2014; Phyu, 2014). Producing plants for the international market requires laboratories and greenhouses, a well-developed infrastructure, and expertise in breeding, growing and marketing plants for export. In our study, reliable internet access and the expertise to develop websites and online commerce also likely played a role. This capacity is well

311 developed in those countries with existing horticultural industries (ITC, 2014) but
312 limited in those such as Laos, where most plants in trade are wild-sourced (Vernon,
313 pers. comm 2014) and only one company was in the early stages of producing
314 orchids legally for trade in 2009 (Lamxay, 2009). Similarly, in 2014 Cambodia had
315 only one well-established nursery, which grew hybrids to supply local cut-flower
316 markets (Jancloes, pers. comm. 2014).
317
318 Although, in theory, building capacity for countries to trade in their own orchids may
319 be a good solution to tackling ABS inequities, this may have negative conservation
320 outcomes. Whilst there are examples of the development of legal trade successfully
321 reducing wild collection (Entwistle et al., 2002) ,there are others showing that
322 demand for wild-sourced products remains stable (Drury, 2009; Dutton et al., 2011),
323 including for the Southeast Asian orchid *Rhynchostylis gigantea* (Phelps et al.,
324 2014). Further, cultivation can also increase wild collection (Williams et al., 2014),
325 and legitimizing trade may facilitate laundering of wild products (Lyons & Natusch,
326 2011), a problem already occurring in the orchid trade as a method to bypass CITES
327 rules (Hinsley et al., 2016b). In addition to these conservation concerns, the CBD
328 recognizes ABS at a state level, giving no guarantee that direct payments or capacity
329 building efforts would reach places where they would benefit development or
330 conservation (Richerzhagen, 2011). People in rural communities may rely on the
331 income from collecting wild animals or plants for trade (Broad et al., 2001), and the
332 development of formal trade may shift profits from these people to a few wealthy
333 business owners (Lybbert et al., 2002; Roe et al., 2002). Where a community
334 approach is taken, as was the case of the appetite suppressant *Hoodia*, it is
335 essential that participants in capacity building projects are not given unrealistic

336 expectations that trade will be an easy, risk-free source of income (Vermeylen,
337 2007).
338
339 Considering these limitations of traditional ABS approaches for the horticultural
340 market, we suggest a different approach to capacity building, one that focusses on
341 strengthening the ability of countries to better control the commercialization of their
342 species. The primary way of doing this is though CITES, which maintained a neutral
343 position on ABS in the past (Roe et al. 2002), but which has developed closer links
344 with the CBD in recent years, including joint meetings in 2016 (Secretariat of CITES
345 and the CBD, 2016). We show that most of the recently described endemic species
346 in trade outside their country of origin have crossed international borders without
347 reported CITES exports, including four CITES Appendix I species. All international
348 movement of orchid species must have CITES paperwork, with some exemptions for
349 trade in seeds, and seedlings in sterile flasks (CITES, 2013). It is possible that some
350 species with no reported CITES exports may have been legally exported as these
351 exempt products, although trade in orchid seed is rare, and the production of
352 seedlings in sterile flasks requires expertise and equipment for propagation. In some
353 cases, this seems unlikely; *Bulbophyllum coweniorum*, a Laotian endemic species
354 with no reported CITES exports, has been popular in trade since at least 2007
355 (Cockel, 2013) but was not being propagated in Laos at this time (Lamxay, 2009). In
356 addition, some endemic species may have been exported with CITES permits that
357 were not reported to CITES by Parties, or low botanical capacity could mean that
358 some endemics are, in fact, naturally present in the neighboring countries where
359 they are being sold. However, this is unlikely to be the case for all the species we
360 identified, and several are likely to have left their country of origin without the correct

361 CITES permits. This is supported by recorded examples of this occurring, most
362 recently in the case of *Paphiopedilum rungsuriyanum,* a Laotian endemic that was
363 first described from a plant that had been wild-collected and transported to a Thai
364 nursery (Gruss et al., 2014). Our findings therefore support those of recent studies
365 showing that the current CITES rules for orchids are not always followed (Phelps &
366 Webb, 2015; Hinsley et al. 2016b).
367
368 To address these problems we suggest capacity building in two key areas. Firstly, to
369 enhance in-country expertise and knowledge of native species by building botanical
370 capacity, which for Cambodia, Laos and Myanmar are amongst the lowest in the
371 region (Seidenfaden, 1992; Schuiteman & de Vogel, 2000). Species often enter
372 trade very quickly after discovery, due to consumer preferences for novelty in these
373 specialist markets (Courchamp et al., 2006; Hinsley et al. 2015). This is especially
374 true in the horticultural trade, where market saturation for commonly traded species
375 has increased the importance of the rapid development of products from new wild
376 species or varieties (Volk & Richards, 2011). Therefore, improving botanical capacity
377 may increase the chances that species are discovered before they have already
378 entered trade and become threatened by over-collection, both of which are common
379 occurrences (Vermeulen & Lamb, 2011; Vermeulen et al., 2014). This may allow
380 conservation measures to be put in place before over-collection occurs, where there
381 is the will and capacity to do so. Secondly, building the capacity of these countries to
382 monitor and control the wild collection and export of their species is also important.
383 This includes strengthening protection of wild plants from over-exploitation and
384 improving the ability of customs officers to detect and identify plants leaving the
385 country. Encouraging CITES Parties to report exports of their orchid species would

allow emerging trade trends to be better monitored, for example via the CITES Review of Significant Trade process. This could be facilitated by efforts to raise the profile of orchid trade in CITES discussions, and increase awareness amongst countries of the value of their native orchid species.

Acknowledgements

T. Lee for advice on network analysis. E.J. Milner-Gulland and Z. Davies for their comments on a draft of this manuscript. The authors gratefully acknowledge the support of the Oxford Martin Programme on the Illegal Wildlife Trade

Financial support

AH was funded by the Natural Environmental Research Council. Grant number: NE/J500458/1

Conflict of interest

None.

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