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Synthesis

Status and conservation of dholes in Indonesia

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Abstract

The dhole (*Cuon alpinus*) is the only canid species native to Indonesia and also the only pack-living large carnivore. It is classified as Endangered by the IUCN with numbers believed to be decreasing throughout their range, yet dholes have received very little scientific and conservation attention compared to other large carnivores in Indonesia. Knowledge on dhole ecology and distribution in Indonesia is limited and has resulted in an inadequate fundament for dhole management and conservation planning for this unique carnivore. This study synthesises existing published data on dholes in Indonesia to assess its conservation status through a "status–pressure–response" framework. Based on dhole presence records published from 1990–2020 we created a distribution map and compared it to a habitat suitability model for dholes developed by the CSG Dhole Working Group. Relative abundance indexes from nine locations in Indonesia are discussed, as are potential threats and knowledge gaps that warrant immediate research priority to support effective conservation planning and intervention of dholes in Indonesia.

Introduction

Habitat loss and fragmentation resulting in loss of connectivity between potentially primary habitats, have led to population declines in many mammalian species (Ceballos and Ehrlich 2002), and is a major threat to many wildlife species across the world (Chapin et al. 2000, Ripple et al. 2014). A combination of large body size, high metabolic demands, and a slow reproductive rate make large carnivores both wide-ranging and rare (Carbone et al. 1999, Woodroffe and Ginsberg 1998). These traits, combined with increasing human densities, make carnivores particularly vulnerable to extinction (Cardillo et al. 2005, Woodroffe 2000). Human dominated landscapes without linkages to suitable habitats, depleted prey bases, and persecution threatens large carnivores globally (Wolf and Ripple 2016, Crooks et al. 2011). Because large carnivores play a critical role in limiting the size of herbivore populations (Dobson et al. 2006, Ripple et al. 2014), the eradication of carnivores can lead to a situation with too many grazers, overgrazing, and deterioration of ecosystems (Beschta and Ripple 2016, Manning et al. 2009, Ritchie et al. 2012).

Indonesia is the world's fourth most populous country and has one of the fastest growing populations with an increase from 90 million in 1961 to 273 million in 2020 (FAO 2020, United Nations 2019). Since 2010, Indonesia's urban population exceeded that of the rural (FAO 2020) and agricultural lands are being converted into urbanised areas in response to the population growth (Firman 1997). Java island accounts for only 9% of the country's total land area but hosts 60% of Indonesia's population, making it one of the most densely populated regions in the world (Liu and Yamauchi 2014). Due to its high human population density and intensive agriculture, Java produces half of the country's rice (Verburg and Bouma 1999, Widiatmaka et al. 2016). Sumatra has much lower human densities (Liu and Yamauchi 2014) and is considered one of the global biodiversity hotspots (Myers et al., 2000). Nevertheless, deforestation and poaching are major threats to wildlife in Sumatra (Margono et al. 2012, Risdianto et al. 2016), although Sumatra still holds large areas of land abundance (Liu and Yamauchi 2014). Despite the high human density, Indonesia is the second most biodiversity rich country in the world, after Brazil, and is considered a high conservation priority (Cincotta et al. 2000, Myers et al. 2000). Indonesia has 733 protected areas spread over 231,946 km^2 covering 12.17% of the total land area, a majority of them found on the islands of Borneo and Sumatra (UNEP-WCMC and IUCN 2020).

The dhole (*Cuon alpinus*) — also known as Asiatic wild dog — is a large (12–20 kg), pack-living canid that occupies a wide range of habitats, from tropical forest and grassland to alpine steppe (Wilson et al. 2009). As opportunistic hunters and scavengers, dholes' diet includes a range of small prey species, such as rodents, beetles, and birds although their main prey consists of medium to large ungulates (Hayward et al. 2014). Packs can vary

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in size from a few to over 30 animals but are most frequently recorded in groups of 5–10 individuals (Durbin et al. 2004). Historically, dholes were widespread across southern and eastern Asia but have now disappeared from over 80% of their former range and are listed as Endangered on the IUCN Red List (Kamler et al. 2015, Wolf and Ripple 2017). Currently, the IUCN Canid Specialist Group estimates that 949–2,215 adult dholes survive globally (Kamler et al. 2015), with only few local population estimates from India (Srivathsa et al. 2021, Selvan et al. 2014) and Thailand (Ngoprasert et al. 2019). A continuous decline in their distribution due to habitat loss, combined with the disappearance of suitable prey, persecution, disease, and possibly interspecific competition threaten this Endangered canid (Kamler et al. 2015, Durbin et al. 2004). As a result, dhole populations are now highly fragmented into several small sub-populations throughout their range (Kamler et al. 2015).

In this study we took a "state–pressure–response" approach to assessing the dhole status in Indonesia. We reviewed research published on dholes in Indonesia and used it to 1) assess their distribution range in Indonesia; 2) identify threats to Indonesian dholes; and 3) outline the knowledge gaps that currently impede effective conservation management of dholes.

Methods

We searched for peer-reviewed scientific articles, book chapters, and reports through ISI Web of Science (www.webofknowledge.com) and Google Scholar (www.scholar.google.com) using the key words "cuon alpinus", "dhole", "cuon javanicus", "asiatic wild dog" and "ajag". Hits were sorted afterwards for studies only within Indonesia. Protected areas were found on Protected Planet (UNEP-WCMC and IUCN 2020) and imported to QGIS (QGIS Version 3.22) along with the dhole species distribution model (SDM) from Java and Sumatra based on Kao et al. (2020).

For every reviewed paper, camera trap data (the number of dhole pictures and the total number of camera trap days) were used to calculate the relative abundance index (RAI). Study site and year were also recorded. Presence records from published literature were classified as either anecdotal or with camera trap data. Reports, non-peer reviewed, and peer reviewed publications with dhole presence mentioned but without photographic evidence were considered as "anecdotal data". We used camera trap data from peer reviewed studies that focused specifically on dholes as well as those where dholes were recorded as "by-catch" in studies that focused on other species. We focused only on studies 1990 onwards to ensure that our assessment reflected the current status.

Current distribution and status in Indonesia

Dhole distribution

Data from the Web of Knowledge revealed a total of 165 dhole studies with only nine from Indonesia. Google Scholar resulted in 10 additional dhole studies from Indonesia. These results are presented in study categories that had "dholes as primary focus" or "dholes recorded as by-catch" (Table 1).

From 1990–2020, dholes were recorded in 11 locations in Java (Figure 1). In western Java, dholes were recorded in Ujung Kulon National Park (NP), Papandayan Reserve, Sawal Reserve, Gede Pangrango NP, and Halimun Salak NP (Qodri et al. 2020, Kao et al. 2020, Rahman et al. 2018). Only one location was reported for central Java, namely Mount Slamet (Sulistyadi 2012), whereas dholes were recorded in East Java in Meru Betiri NP, Alas Purwo NP, Baluran NP, Kawah Ijen Nature Tourism Park, and Bromo Tengger Semeru NP (Kao et al. 2020, Iyengar et al. 2005).

On Sumatra, dholes were reported in 14 locations (Figure 2). Ulu Masen Ecosystem, Gunung Leuser NP, and Batang Toru Ecosystem in Northern Sumatra, and in Bukit Tigapuluh NP, Bukit Dua Belas NP, and Harapan Rainforest in central Sumatra, and in Berbak-Sembilang NP, Kerinci-Sebelat NP, Bukit Balai Rejang Forest in southern Sumatra (Kao et al. 2020, Silalahi et al. 2017, Radinal et al. 2019, Durbin et al. 2004). Recent studies using by-catch data from a camera trap study on tigers (*Panthera tigris*) confirmed the presence of dholes in Bukit Rimbang Bukit Baling Wildlife Reserve, Tesso Nilo NP, Bukit Bungkuk Nature Reserve, and Bukit Betabuh Protection Forest (Widodo et al. 2020). Another recent camera trap study confirmed the presence of dholes in the Bukit Barisan Selatan NP (Allen et al. 2020). Figure 1 illustrates published data from 1990–2020 that present camera trap data or anecdotal data with mentions of dhole presence but without empirical data such as camera trap photos.

Many of the presence locations (Figure 1 and 2) are based on anecdotal data rather than empirical data. On Sumatra, four of the areas with anecdotal evidence are protected areas: Bukit Tiga Puluh NP, Bukit Dua Belas NP, Berbak-Sembilang NP, and Kerinci-Seblat NP, with two from non-protected areas; Harapan Rainforest and Batang Toru Ecosystem (Silalahi et al. 2017, Kao et al. 2020, Durbin et al. 2004). Kao et al. (2020) state that all presence records in their SDM model are from either camera traps, sightings, faeces or tracks, however, it is not specified what kind of data is present from each site. Presence evidence from systematic camera trapping studies was found from six protected areas: Gunung Leuser NP, Bukit Bungkuk Nature Reserve, Bukit Rimbang Bukit Baling Wildlife Reserve, Tesso Nilo NP, Barisan Wildlife Reserve, and Bukit Barisan-Selatan NP (Widodo et al. 2020, Allen et al. 2020, van Schaik and Griffiths 1996). Dholes were also recorded with camera traps at three additional locations in non-protected areas: Ulu Masen Ecosystem in northern Sumatra, Bukit Betabuh Protection Forest, and Peranap in central Sumatra (Radinal et al. 2019, Sunarto et al. 2015, Widodo et al. 2020).

On Java, we found empirical evidence of dhole presence from only two national parks: Baluran NP (Pudyatmoko 2017, Nurvianto et al. 2015a) in the north-eastern corner and Ujung Kulon NP in the far west (Rahman et al. 2018). Gunug Gede Pangrango NP has anecdotal evidence (Kao et al. 2020) but Ario et al. (2020) failed to detect dholes during a camera trap study in 2018 in the national park. Anecdotal evidence was also reported from Alas Purwo NP, Bromo Tengger Semeru NP, Gunung Halimun-Salak NP, Gunung Sawal Wildlife Reserve, Kawah Ijen Nature Tourism Park, Meru Betiri NP, Papadayan Nature Reserve, and from one non-protected area; Gunung Slamet (Kao et al. 2020, Iyengar et al. 2005, Indrawan et al. 1996).

Dhole population size

Dholes are social canids that live in packs with up to 30 individuals (Durbin et al. 2004) and with no individually recognisable markings there are currently no reliable methods available to estimate dhole population size across their range (Srivathsa et al. 2020c). However, recent methods have been developed to estimate dhole densities in Thailand and India (Srivathsa et al. 2021, Ngoprasert et al. 2019). Trapping success rate or relative abundance index (RAI) can be used as a proxy for abundance (O'Brien et al. 2003). However, many factors may influence RAI such as habitat type, number of camera trap nights, and type of survey making direct comparison of RAIs across different locations and between species potentially inaccurate (Sollmann 2018). RAI is commonly reported for dholes and many other species in Indonesia and is currently the best available information (Table 1).

In the Ulu Masen Ecosystem, a non-protected area in north Sumatran Aceh Province, a study from 2017 published comparative trapping rates for three large carnivores: Sunda clouded leopard (*Neofelis diardi*) with an RAI of 2.15, tigers with an RAI of 0.45, and dholes with an RAI of 0.37 (Radinal et al. 2019) (Table 2). In Central Sumatra, Riau province in Kampar, Kerumutan Wildlife Reserve, Tesso Nilo NP, Peranap and Bukit Rimbang Bukit Baling NP, Sunarto et al. (2013) found a mean RAI of 0.79 for tigers, whereas dholes had almost five times lower trapping rates with an RAI of 0.16 within the same study site and period from 2005–2007 (Table 2). Dholes had the highest trapping rate in the unprotected forest Peranap, which also had the highest rate of potential prey, whereas no dholes were detected in the lowland peat forests Kampar and Kerumutan (Sunarto et al. 2015) (Table 2). A more recent study from Bukit Rimbang Bukit Baling (BRBB) also provided comparative RAI values for dholes and tigers within the same study site from 2011–2015. In the north-eastern part, RAI scores for dholes and tigers were similar, with 0.53 for dholes and 0.57 for tigers, the highest reported in the study for dholes - but the lowest of all for tigers. The north-western part BRBB recorded a much higher RAI for tigers (2.59) but only 0.32 for dholes, and the southern part had the lowest RAI score for dholes (0.09) and 0.89 for tigers (Widodo et al. 2020; Widodo et al. 2017) (Table 2). In Bukit Barisan Selatan NP in southern Sumatra, tigers had the highest mean annual RAI (2.41), followed by the Asiatic golden cat (*Catopuma temminckii*; 0.88), marbled cat (*Pardofelis marmorata*; 0.6), and Sunda clouded leopard (0.39), whereas dholes had the lowest reported RAI of all carnivores (0.33; Allen et al. 2020). Sumatran tigers are considered Critically Endangered, but in Bukit Barisan Selatan NP their mean annual RAI is more than seven times greater than that for dholes (Allen et al. 2020; Table 2). In Baluran NP, East Java, dholes had the highest reported RAI for any location in Indonesia with 5.38, three times greater than leopards (1.79) within the same study (Pudyatmoko 2017). Ujung Kulon NP in West Java had the second highest RAI score for dholes (0.89), with leopards approximately twice as high (1.77; Rahman et al. 2018; Table 2).

Main threats to dhole conservation

Habitat fragmentation is the major threat to carnivores worldwide because it often results in populations being fragmented into several small and isolated sub-populations that suffer increased risk of inbreeding (Ripple et al. 2014). With large scale deforestation still taking place in Indonesia every year (Margono et al. 2012), existing dhole populations are at risk of becoming increasingly isolated from each other with little chance of interaction. A genetic study using samples from dholes in Baluran NP collected in 1998 showed a low genetic variation, suggesting that this population may already suffer from the effect of isolation due to limited influx of new genetic material (Iyengar et al. 2005).

Decrease in prey is another major threat to dholes throughout their range with 42% of prey species classified as threatened and a decreasing trend for 81% of prey species (Wolf and Ripple 2016). Based on this Wolf and Ripple (2016) argue that dholes are in the top five of large carnivores at particular risk of prey depletion. On Java 50–60% of dhole prey species are estimated to be decreasing while on Sumatra this estimate is 60–80% (Wolf and Ripple 2016).

Only limited information about the impact of logging and other anthropogenic activities exists on dhole ecology. Nurvianto et al. (2015b) found that human encroachment and game hunting may pose a threat to dholes in Baluran NP. However, in another study Pudyatmoko (2017) found that dholes in Baluran NP did not appear to avoid areas with livestock grazing and human settlement. In the study of Sunarto et al. (2015) in the Riau Province, Sumatra, very high logging activities and high human presence (RAI 5.25) corresponded with low dhole camera trap rates (RAI 0.16), but comparatively high rates for tigers (RAI 0.79). Similar camera trap rates of dholes were found for the higher elevation forests of the unprotected Peranap and Bukit Rimbang Bukit Baling Wildlife Reserve, despite higher observed logging activities and human presence (RAI 1.29) in Peranap than in Bukit Rimbang Bukit Baling (human RAI 0.70) (Sunarto et al. 2015). In Peranap no tigers were detected and in Bukit Rimbang Bukit Baling tigers had a low camera trap rate (RAI 0.4). No dholes were detected in the two peat swamps Kampar and Kerumutan (Sunarto et al. 2015). Kampar is not a protected forest and has high logging activity and high human presence (RAI 2.47), whereas Kerumutan is a Wildlife Reserve with low logging activity, low human presence (RAI 0.05), and tiger presence (RAI 0.7) (Sunarto et al. 2015). Sunarto et al. (2015) suggest that dholes might avoid tigers as they generally found lower camera trapping rates of dholes in areas where tigers were present. Competition with other large carnivores could potentially be a threat, particularly if food resources are limited (Wolf and Ripple 2016, Kamler et al. 2015).

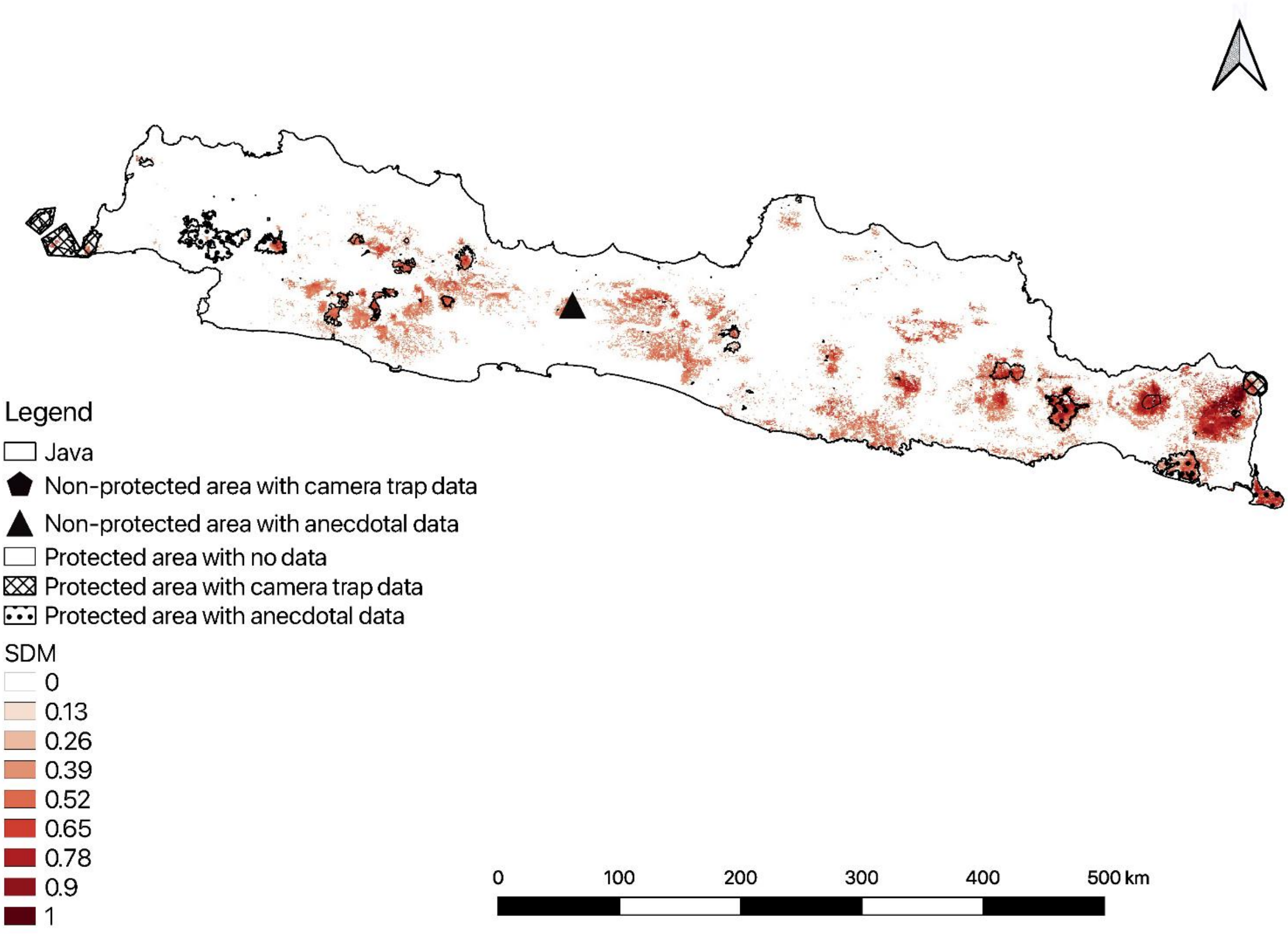

Figure 1. Map of Java illustrating the potential dhole distribution based on Species Distribution Model (SDM) generated by MaxEnt with probability of species presence (environmental suitability) for suitable patches from Kao et al. (2020). Increased redness corresponds to more suitable habitat. Protected areas (PA) are outlined and PAs with camera trap (CT) confirmed presence of dholes are hashed, PA with anecdotal evidence are marked with dots. Non-protected areas (NPA) with CT confirmed presence of dholes are marked with pentagons, NPA with anecdotal evidence of dhole presence is marked with triangles. Presence data is based on published literature between 1990–2020.

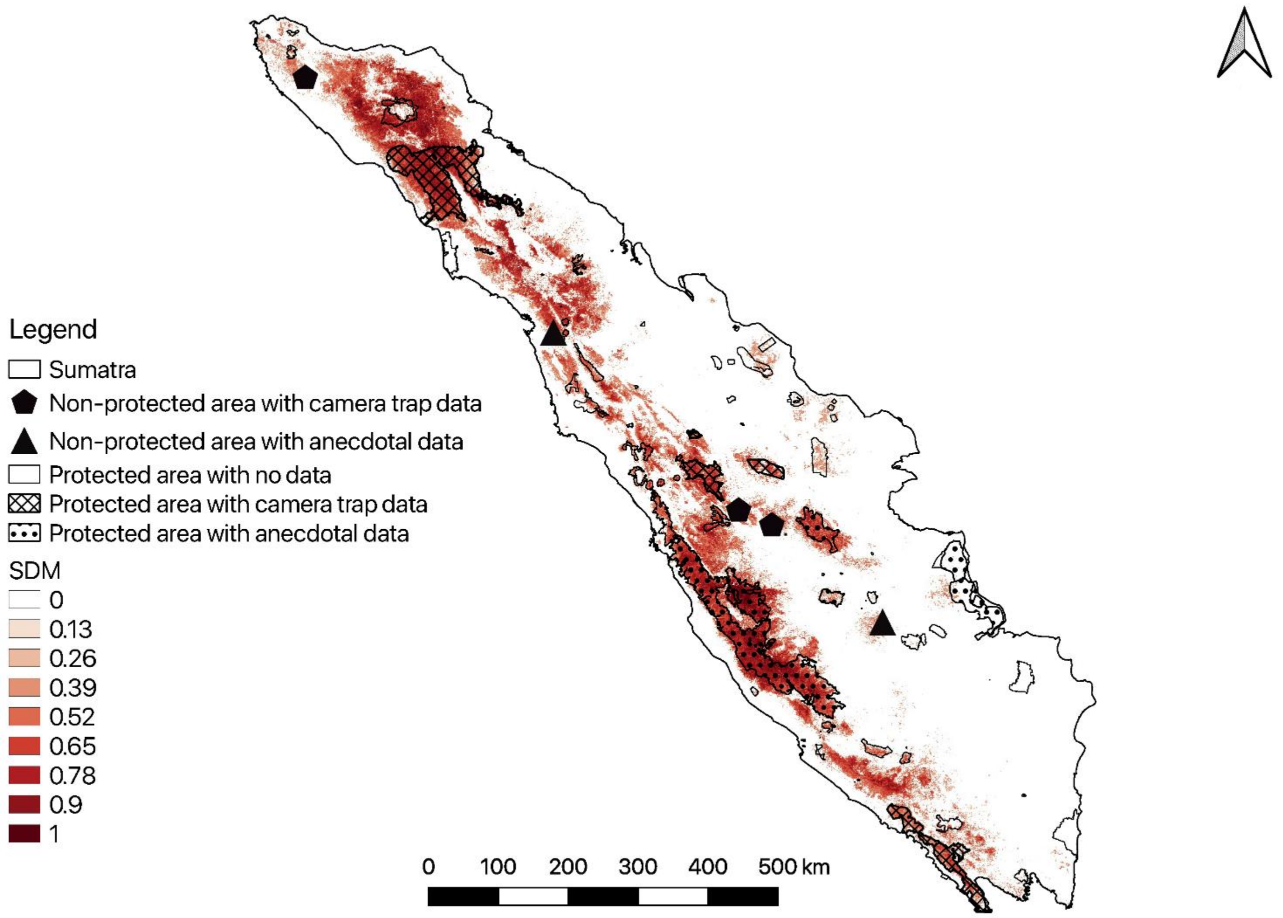

Figure 2. Map of Sumatra illustrating the potential dhole distribution based on Species Distribution Model (SDM) generated by MaxEnt with probability of species presence (environmental suitability) for suitable patches from Kao et al. (2020). Increased redness corresponds to more suitable habitat. Protected areas (PA) are outlined and PAs with camera trap (CT) confirmed presence of dholes are hashed, PA with anecdotal evidence are marked with dots. Non-protected areas (NPA) with CT confirmed presence of dholes are marked with pentagons, NPA with anecdotal evidence of dhole presence is marked with triangles. Presence data is based on published literature between 1990-2020.

Retaliatory killing due to livestock predation is of great concern to dhole conservation in Bhutan (Katel et al. 2015, Wang and Macdonald 2006), Nepal (Aryal et al. 2015), and India (Lyngdoh et al. 2014). However, in Indonesia there is no published literature on retaliatory killings of dholes due to livestock predation. However, reports in Bahasa Indonesia from Baluran NP reports retaliatory killings of dholes, their pups and destructions of dens occur after predation events (Dwiputra 2015). Predation of livestock, which grazes illegally inside Baluran NP (Pudyatmoko et al. 2018), has been directly observed on multiple occasions (Nurvianto et al. 2016, Dwiputra 2015). The conflict between dholes and livestock herders has likely been ongoing for decades and is probably widespread throughout Indonesia but is poorly recorded due to lack of reporting and studies (Dwiputra 2015). Dholes also prey on banteng (*Bos javanicus*) which complicates conservation intervention, because both species are extremely rare and listed as Endangered on the IUCN Red-list and as Totally Protected Species in Indonesia (Nurvianto et al. 2016, Macdonald 2004). Dholes alone have been accused for the decline of banteng on Java (Hedges and Tyson 1996, Pudyatmoko et al. 2007). Especially dholes in large packs are believed to pose a threat to the long-term survival of banteng according to Pudyatmoko et al. (2007). To alleviate the supposed dhole predation, Pudyatmoko et al. (2007) suggested dhole pack sizes should be limited as a conservation measure to protect banteng. However, the effects on the social dynamics and hunting success of reducing dhole pack size remain unknown. Dhole hunting success may be dependent on a critical minimum pack size, as has been found in African wild dogs (*Lycaon pictus*) (Courchamp and Macdonald 2001).

Disease transfers from domestic animals and feral predators pose additional risks to wildlife (Hughes and Macdonald 2013). The degree of disease transmission between domestic/feral animals and wildlife remains unknown in Indonesia, but disease transmission from domestic dogs to wild canids have had severe consequences for several other species (Berentsen et al. 2013, Johnson et al. 2010, Randall et al. 2004, Woodroffe et al. 2004). This can potentially be one of the most significant risks to, especially Javan dholes, due to its high degree of interaction with and overlapping habitat use with humans. Both retaliatory killings due to loss of livestock and risk of disease transfer may become more widespread with declining populations of wild prey as it might force dholes closer to humans in search for food (Berger et al. 2013).

In summary, many of the drivers i.e., habitat loss, poaching, prey decline and potentially competition with sympatric carnivores are known, however, to what extent and the exact effect these drivers have on dhole population decline in Indonesia remains poorly understood. Unlike the attention afforded to other large carnivores (e.g., leopards and tigers), the population ecology and conservation priorities of dholes in Indonesia is in urgent need of more attention.

Discussion

Historically, dholes were distributed throughout Java and Sumatra (Durbin et al. 2004). However, information from our literature review suggests that the dhole distribution extent in Indonesia by 2020 has decreased dramatically to 25 isolated sites. Only a few studies have focused on dhole ecology in Indonesia. These include activity patterns (Allen et al. 2020, Widodo et al. 2020, Rahman et al. 2018, Nurvianto et al. 2015a), spatial partitioning with sympatric carnivores and/or prey species (Rahman et al. 2018, Pudyatmoko 2018) and Iyengar et al. (2005) investigated dhole phylogeography across their range. In Baluran NP, two dhole studies focused on diet and the effect of human/prey interaction respectively (Nurvianto et al. 2016, Nurvianto et al. 2015b). Prey availability, water sources and human disturbance are generally considered important factors affecting carnivores including dholes (Nurvianto et al. 2015b, Steinmetz et al. 2013, Srivathsa et al. 2017). However, more knowledge is needed to get reliable population estimates in Indonesia and identify threats.

The concept of "umbrella species" is widely acknowledged, although it is not always possible to rely on other species to function as umbrella species to ensure conservation of all species within an ecosystem (Li et al. 2020). In the case of dholes, it is necessary to monitor dhole population dynamics throughout their range to ascertain if measures taken to protect other carnivores e.g., tigers and leopards also has a positive effect on dholes or, if different conservation interventions are required to ensure their long-term survival (Kumar et al. 2019, Srivathsa et al. 2020c). Sunarto et al. (2015) suggested that dholes avoid tigers, hence using camera trap data from tiger studies to estimate dhole population density may be misleading and protecting tiger habitat cannot necessarily double up as protecting dhole habitat too. Very different requirements may be needed for solitary carnivores like tigers to recover (Karanth et al. 2020) compared to social predators with cooperative hunting and breeding (Courchamp and Macdonald 2001). In addition, the effort to get camera trap photos of rare species may take longer than common and abundant species (Thompson and Withers 2003). Ario et al. (2020) failed to detect dholes in Gunung Gede Pangrango NP during a camera trap study in 2018. However, the study only consisted of 623 camera trap days so rare and elusive animals like dholes may not have been detected within the limited timeframe although leopards were detected on four occasions.

Java as it is one of the most densely human populated islands in the world (Dibia et al. 2015), which has undergone widespread and rapid landscape changes over the last few decades, converting large areas of natural habitat to agricultural land (Sodhi et al. 2010). Little undisturbed habitat is left for wildlife (Nurvianto et al. 2015b, Iyengar et al. 2005) with protected areas isolated from each other throughout Java (Kamler et al. 2015). However, dholes' ability to move between protected areas may play an important role in future conservation management, and studies on dhole presence and habitat utilisation outside PAs are highly needed. On Sumatra Sunarto et al. (2015) found dholes in a non-protected area in Riau Province and it is possible that Sumatra still holds undocumented dhole populations.

According to the IUCN Red List India holds the largest population of dholes in the world, followed by Thailand and Myanmar supporting smaller populations (Kamler et al. 2015). No population estimate is currently available for Indonesia (Kamler et al. 2015), but Srivathsa et al. (2020b) assumes that the Indonesian dhole population is very small. However, Sumatra has a few large national parks where dholes could potentially still exist in greater numbers. These large protected areas may still harbour a substantial population of dholes and could prove to be important core areas for dholes in Indonesia. One such case is Indonesia's largest national park Kerinci-Seblat which covers some 13,754 km^2 (UNEP-WCMC and IUCN 2020). Despite high poaching rates (Rayan and Linkie 2016), illegal logging (Linkie et al. 2003), deforestation and conversion to farmland (Linkie et al. 2007, Hariyadi and Ticktin 2012) within Kerinci-Seblat NP, tigers still occupy 83% (Wibisono et al. 2011). However, there is no published data on dholes from Kerinci-Seblat NP and it remains unknown if they are currently present.

Table 1. Relative Abundance Index (RAI) of dholes (*Cuon alpinus*) in Indonesia. RAI is calculated as the number of independent events (>30 mins between photos)/100 camera trap days.

Location	Study period	Independent events	Camera trap days	RAI	Original study focus	Reference
Baluran National Park	Aug 2015–Jan 2016	84	1562	5.38	Terrestrial community	Pudyatmoko 2017
Bukit Barisan Selatan National Park	Apr 2010–July 2017	4	11896	0.03	Terrestrial community	Allen et al. 2020
Bukit Betabuh Protected Forest	Jan–April 2013	3	1791	0.17	Tiger	Widodo et al. 2020
Bukit Bungkuk Nature Reserve	June–Sep 2012	8	1762	0.45	Tiger	Widodo et al. 2020
Bukit Rimbang Bukit Baling Wildlife reserve	Nov 2011–Dec 2015	22	8125	0.27	Tiger	Widodo et al. 2020
Riau Province, Central Sumatra*	May 2005–Nov 2007	12	7513	0.16	Tiger	Sunarto et al. 2015
Tesso Nilo National Park	July–Nov 2013	4	2335	0.17	Tiger	Widodo et al. 2020
Ujung Kulon National Park	Jan–Dec 2013	351	39420	0.89	Javan Rhino	Rahman et al. 2018
Ulu Masen Ecosystem	Apr–Sep 2017	14	3740	0.37	Terrestrial community	Radinal et al. 2019

*Kampar, Kerumutan Wildlife Reserve, Peranap, Bukit Rimbang Bukit Baling NP, Tesso Nilo NP.

Table 2. Relative abundance index (RAI) for dholes (*Cuon alpinus*) and sympatric large carnivores in Indonesia with tigers (*Panthera tigris*) in Sumatra and leopards (*Panthera pardus*) in Java. Bukit Rimbang Bukit Baling Wildlife Reserve is both for the total area and split into three regions in the Northeastern, Northwestern and Southern.

Location	Study period	Tiger RAI	Leopard RAI	Dhole RAI	Reference
Baluran National Park	Aug 2015–Jan 2016		1.79	5.38	Pudyatmoko 2017
Bukit Barisan Selatan National Park	2010–2017	0.24		0.03	Allen et al. 2020
Bukit Rimbang Bukit Baling Wildlife Reserve	Nov 2011–Dec 2015	1.49		0.27	Widodo et al. 2017, Widodo et al. 2020
Northeastern	Nov 2011–Feb 2012	0.57		0.53	Widodo et al. 2017, Widodo et al. 2020
Northwestern	Feb–June 2014	2.59		0.32	Widodo et al. 2017, Widodo et al. 2020
Southern	Aug–Dec 2015	0.89		0.09	Widodo et al. 2017, Widodo et al. 2020
Riau Province, Central Sumatra*	May 2005–Nov 2007	0.79		0.16	Sunarto et al. 2013, Sunarto et al. 2015
Tesso Nilo National Park	Apr–July 2008 July–Nov 2013	4.50		0.17	Sunarto et al. 2013, Widodo et al. 2020
Ulu Masen Ecosystem	Apr–Sep 2017	0.45		0.37	Radinal et al. 2019
Ujung Kulon National Park	Jan–Dec 2013		1.77	0.89	Rahman et al. 2018

*Kampar, Kerumutan, Peranap, Bukit Rimbang Bukit Baling, Tesso Nilo National Park

Alas Purwo, Meru Betiri and Baluran are neighbouring national parks in East Java, where dholes have been reported (Durbin et al. 2004, Iyengar et al. 2005). The three national parks are situated within ~100 km of each other and loosely connected by the Belambangan Biosphere Reserve (UNESCO 2016). It remains unknown if dholes make use of the corridor to disperse between these national parks (Iyengar et al. 2005). Dhole dispersal between these protected areas is critical for preservation of the genetic variability in the east Javan dhole population(s) (Iyengar et al. 2005, Robert 2009).

Based on the SDM results from Kao et al. (2020), Baluran NP and Ujung Kulon NP have relatively low suitability (Figure 1) but both areas have high RAI values recorded for dholes (Table 1). The SDM model builds on predictors including bioclimatic factors (e.g., mean diurnal temperature range and annual precipitation), tree cover, land cover, human footprint, elevation, aspect, slope, ruggedness and human population density (Kao et al. 2020). On Java, this means that the most suitable habitats turn out to be on the upper slopes of volcanos (Figure 1) with low anthropogenetic impact, high precipitation and more tree cover than most other areas. However, dholes in Java are also known to occur in open habitat (Pudyatmoko 2017) with the highest reported camera trapping rate found in Baluran NP which contains a mosaic of habitats including the largest remnant of the Sundaland savannah (Iyengar et al. 2005). The accuracy of SDMs is limited to the quality of data and predictor variables included in the model (Guillera-Arroita et al. 2015, Brodie et al. 2020). Prey availability is considered one of the most important factors for carnivores (Wolf and Ripple 2016, Karanth et al. 2004) but was not included in the SDM from Kao et al. (2020). In an occupancy probability model from India a wild prey index was incorporated and turned out to be one of the most important predictors for dhole occupancy (Srivathsa et al. 2020a). Open habitats such as savanna and savanna woodlands generally support more large prey than closed habitats such as tropical forests (Fritz and Loison 2006, Cavada et al. 2019). A high level of prey biomass in Baluran NP could explain the high trapping rate. Although not directly comparable, the reported RAI for Baluran NP (5.38) is higher than that for Bandipur Tiger Reserve (RAI=3.5) in Southern India which is considered to have high densities of dholes (Karanth et al. 2017). From western Thailand Khoewsree et al. (2020) reported an RAI of 7.41 for dholes in Khao Yai National Park, both areas scored a high probability of species presence in the same SDM (Kao et al. 2020). However, both RAI and MaxEnt-based suitability models are likely to be somewhat inaccurate.

For Sumatra the SDM from Kao et al. (2020) might be more representative (Figure 2), a distribution model from Rimbang Baling and Tesso Nilo NP found that forests cover contributed with 83% and is therefore believed to be the most important factor for dholes in Sumatra (Widodo et al. 2020). Although all areas in Sumatra sampled by camera traps had relatively low RAI scores compared to Java (Table 1).

Conclusions

We urge managers, scientists, and students working on dholes and their prey to publish findings and encourage more studies specifically targeting dholes and their prey in protected areas across the Indonesian range. Areas categorised by Kao et al. (2020) as "high suitability" are of special interest. Several of these are already protected with empirical or anecdotal evidence for dhole presence since the 1990s such as Gunung Leuser NP (7927 km^2) in northern Sumatra, that has camera trap data from 1987–1991. These data are 30 years old and a validation of the presence of dholes there is important for the development of effective conservation and management interventions. Kerinci-Seblat NP (13,754 km^2) and Bukit Tiga Puluh NP (1,277 km^2) in central Sumatra should also be considered priority areas because both are assessed as high environmental suitability according to the species distribution model (Kao et al. 2020). Anecdotal evidence of dhole presence exists from Meru Betiri NP (580 km^2) and Alas Purwo NP (434 km^2) with observations from 1993 and 1994 (Indrawan et al. 1996), but no camera trap studies have been published to verify their presence since. Most studies have used camera traps to verify dhole presence but new methods e.g., environmental DNA has proven successful at detecting terrestrial mammals from waterholes (Seeber et al. 2019, Wilcox et al. 2021) and may be used as an alternative method. Several studies have focused on dholes in Baluran NP that is connected to Alas Purwo NP and Meru Betiri via Kawah Ijen Nature Reserve together forming the Belambangan Biosphere Reserve (UNESCO 2016). However, whether dholes are able to move through this corridor remains unknown. The SDM model (Kao et al. (2020) predicted Bromo Tengger Semeru NP (503 km^2) and Gunung Gede Pangrango NP (220 km^2) as "highly suitable" and have past anecdotal evidence of dhole presence but no post-1990 published data to confirm since then. Assessing to what degree Indonesia's dholes exist in small, isolated, and genetically vulnerable subpopulations is critical for developing a meta-population management plan for Java.

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