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Policy analysis

The scale of biodiversity impacts of the Belt and Road Initiative in Southeast Asia

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ABSTRACT

The Belt and Road Initiative (BRI) is the largest infrastructure development in human history. Given its scale of influence and infrastructure undertakings, it is set to bring far-reaching environmental impacts to regions such as Southeast Asia, one of the biologically richest and most diverse regions in the world. Knowing where and what biodiversity BRI will potentially affect is crucial to plan and address its negative impacts. Using BRI transport infrastructure spatial data, we conducted a GIS analysis of the potential BRI impacts in Southeast Asia on terrestrial and marine biodiversity indicators, including protected areas (PAs), Key Biodiversity Areas (KBAs), terrestrial ecoregions, forest cover, threatened species, and fragile ecosystems such as seagrasses, mangroves, and coral reefs. We assessed the potential impacts across four key distance thresholds (1, 5, 25, and 50 km "impact zones") on either side of the routes. For the terrestrial routes we assessed impacts for five different types of linear rail and road infrastructure development. Within 1 km of all routes 32 PAs, 40 KBAs and 29 ecoregions are intersected. While, 142 threatened species including 26 critically endangered species are within 5 km from new rail, which are also commonly found in frontier landscapes. In marine ecosystems 20 marine PAs and 16 KBAs are intersected by BRI marine routes. We conclude by discussing ways BRI could minimise its environmental impacts and utilise its political weight to advance conservation efforts in host nations.

1. Introduction

Many of the low and middle income countries are currently witnessing an unprecedented expansion of infrastructure, such as roads, railways, power lines, gas lines, canals, settlement, utilities and dams (Laurance et al., 2009, 2015a). The ever growing pursuit for natural resources such as timber, oils and minerals, arable land, as well as initiatives to increase regional trade, transportation, and energy infrastructure are among the driving forces of the worldwide proliferation of infrastructure (Laurance et al., 2014) and consequent decline in biodiversity (Sloan et al. 2016). Infrastructure development and in particular linear infrastructure such as roads and railways causes the decline and extinction of wildlife populations in terrestrial ecosystems through habitat loss, degradation, and fragmentation (Clements et al., 2014; Hughes, 2018; Laurance et al., 2015a; Torres et al., 2016). Fragmentation from linear infrastructure has a number of negative effects such as the isolation of remnant habitat, increased wildlife mortality from roadkills, facilitation of biological invasions, accelerated forest conversion and increased illegal activities such as poaching, due to easier accessibility, as well as illicit exploitation of natural resources (Forman and Alexander, 1998; Laurance et al., 2009; Raman, 2011). Frontier regions rich in biodiversity (Sloan et al., 2019a, 2019b) which provide ecosystem services at global, regional, and local scales may be particularly vulnerable to such expansion.

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Table 1
Belt and Road (BRI) projects. Adapted from Reed and Trubetskoy (2019). CICPEC = China-Indochina Peninsula Economic Corridor; BCIM = Bangladesh-China-India-Myanmar Economic corridor.

Route type with total length in brackets	BRI corridors	Projects	Project improvements	Length (km)
New road (226 km)	CICPEC	Sihanoukville Port	Phnom Penh-Sihanoukville road	226
New high capacity rail (1305 km)	CICPEC	Kuala Lumpur-Singapore HSR	Kuala Lumpur-Seremban-Singapore rail	338
	BCIM	Kunming-Calcutta HSR	Kunming-Calcutta rail	967
New rail (1926 km)	BCIM	Dali-Lashio rail	Dali-Lashio rail	116
	BCIM	Kalay-Tamu-Jiribam rail	Kalay-Tamu-Jiribam rail	121
	CICPEC	MRL East Coast Rail Link	Kuala Lumpur-Pahang-Kelantan rail	565
	CICPEC	Kunming-Vientiane rail	Kunming-Vientiane rail	472
	CICPEC	Burma Railway	Nam Tok-Thanbyuzayat rail	189
	CICPEC	Vietnam-Cambodia rail	Phnom Penh-Ho Chi Minh rail	224
	CICPEC	Sihanoukville Port	Phnom Penh-Sihanoukville rail	239
Road upgrade (1205 km)	BCIM	Kyaukpyu Port	Kyaukpyu-Mandalay road upgrade	661
	CICPEC	Noi Bai-Lao Cai expressway	Noi Bai-Lao Cai road	319
	CICPEC	Highway AH3	Xishuangbanna-Huay Xai road	225
Upgrade to high capacity rail (4475 km)	CICPEC	Bangkok-Pedang Besar-Kuala Lumpur rail	Bangkok-Pedang Besar-Kuala Lumpur rail	1712
	CICPEC	Bangkok-Rayong HSR	Bangkok-Rayong HSR	217
	CICPEC	Bangkok-Vientiane rail	Bangkok-Vientiane rail	637
	CICPEC	Gemas-Johor Bahru rail	Gemas-Johor Bahru rail	194
	CICPEC	Vietnam National HSR	Hanoi-Ho Chi Minh City rail	1715

Emerging as the world's second largest economy, in 2013, Chinese President Xi Jinping launched an ambitious foreign policy known as the Belt and Road Initiative (BRI) (also known as One Belt One Road) (Communist Party of China (CPC) news, 2013). It is a regional economic development initiative inspired by the ancient Silk Road which aims to promote economic cooperation and facilitate trade growth between China and various nations across the world. BRI consists of a web of infrastructure megaprojects such as roads, expressways, high speed rails, pipelines, ports and power plants, along two main routes, the land-based Silk Road Economic Belt and the marine-based 21st Century Maritime Silk Road which span across >71 countries (Belt and Road Portal, 2017). With Chinese investments scaling up to USD 4–8 trillion, the BRI is potentially the largest infrastructure development initiative in the 21st Century, encompassing 30% of the world's GDP, 62% of world's population and 75% of currently known energy reserves (The World Bank, 2018), dwarfing previous development undertakings. While the BRI will potentially provide positive socio-economic benefits to BRI countries, it is expected to have far reaching environmental impacts through areas it traverses and potentially will have “significant consequences for biodiversity” (Lechner et al., 2018).

Only recently has the conservation literature drawn attention to potential biodiversity impacts associated with the BRI (Ascensão et al., 2018; Foggin et al., 2018; Hughes, 2019; Lechner et al., 2018; Liu et al., 2019; Teo et al., 2019). A key concern for conservation is which BRI corridors overlap with biodiversity hotspots such as those across Southeast Asia. Southeast Asia is among the most biodiverse regions in the world, home to four of the Earth's 34 biodiversity hotspots and supporting a large number of endemic species (de Bruyn et al., 2014). In addition Southeast Asia also has high marine biodiversity for corals, seagrass beds and mangrove forests and is home to the Coral Triangle (Lechner et al 2018) often described as the Amazon of the ocean. The region contains >600 of the nearly 800 reef-building coral species found worldwide, houses approximately 35% of the world's mangroves species and over 45% of the seagrass species (Burke et al., 2002).

In Southeast Asia BRI will manifest as an overland China-Indochina Peninsula Economic Corridor (CICPEC) and the Bangladesh-China-India-Myanmar Economic corridor (BCIM-EC). Maritime routes will travel through the South China Sea to the Andaman Sea via the Straits of Malacca (Sims and Pinto, 2019). With ASEAN countries partnering with China to enhance regional economic growth as well as promote close regional trade with China, Southeast Asia is set to become a hotspot for booming infrastructure development. BRI-related infrastructure will traverse through Southeast Asia via various transportation corridors that, if poorly planned, could pose detrimental impacts to ecosystems and biodiversity in the region (Lechner et al., 2018). Concerns regarding BRI projects in the region are primarily economic and geo-political (Liu and Lim, 2018), while their environmental implications are poorly known and have received little attention so far (Ascensão et al., 2018; Hughes, 2019; Lechner et al., 2018; Teo et al., 2019).

The aim of this study is to assess the potential scale of BRI impacts on marine and terrestrial biodiversity in Southeast Asia. Southeast Asia already represents one of the most threatened and overlooked regions globally (Sodhi et al., 2010; Hughes, 2017a, 2017b; Morand et al., 2017) and the expected expansion of infrastructure and trade across this region is likely to exacerbate many of the existing threats to biodiversity in the region. Here we mapped the expected BRI routes and quantified the proportional area of important ecosystems and concentrations of threatened species in the vicinity of BRI routes at varying distances from the terrestrial linear transport and marine sea routes. For the terrestrial routes, we also assessed potential impacts for five different types of linear infrastructure development, each of which are likely to differ in the intensity of impacts on biodiversity: new road, new high capacity rail, new rail, road upgrade, and upgrade to high capacity rail. We identified conservation priority regions where planned BRI routes and protected areas overlap with expected BRI impact zones. We then assessed the relationship between BRI routes and human footprint and forest cover in the region to assess whether BRI routes occur within frontier/natural landscapes rather than areas of existing human disturbance. Finally, we discuss ways in which BRI may mainstream biodiversity conservation into its policy to mitigate potential negative impacts while satisfying the need for development in the Southeast Asian region. The paper also includes detailed data on specific impacts such as individual IUCN listed species and protected areas that will potentially be affected.

2. Materials and methods

2.1. Data sources and pre-processing

We obtained roads and rails spatial data from a transport infrastructure database supplied by a World Bank database associated with a report by Reed and Trubetskoy (2019) (https://datacatalog.worldbank.org/dataset/bri-database-reed-and-trubetskoy-2019) (Tables 1, S1) and used in previous World Bank BRI reports (Baniya et al., 2019; Losos et al., 2019). From that database we identified roads and rails planned or constructed under the BRI and also validated and compared those routes to MERICS (2018) (perhaps the most widely shared BRI route

Table 2
Ecological datasets used in the analysis.

Dataset	Data type	Data source	Feature classes	Purpose
Terrestrial				
2009 Human footprint	Raster	Venter et al. (2016)	Human footprint classes	Assess intactness
Forest cover	Raster	Li et al. (2016)	Forest cover	Assess intactness
Terrestrial Ecoregions of the World	Vector	Dinerstein et al. (2017)	Ecoregion names, Nature Needs Half	Assess vulnerability
Protected area	Vector	IUCN and UNEP-WCMC, 2018	PA names, IUCN categories, countries	Assess vulnerability
Key biodiversity areas	Vector	BirdLife International, 2019	KBA names, countries	Assess vulnerability
Terrestrial biodiversity				
IUCN Bird range	Vector	BirdLife International and Handbook of the Birds of the World, 2018	Species names, IUCN categories	Assess vulnerability
IUCN Terrestrial mammal range	Vector	IUCN 2019	Species names, IUCN categories	Assess vulnerability
IUCN Amphibian range	Vector	IUCN 2019	Species names, IUCN categories	Assess vulnerability
GARD Reptile range	Vector	Roll et al., 2017	Species names, IUCN categories	Assess vulnerability
Marine				
Coral reef distribution	Vector	UNEP-WCMC 2018	Coral reefs	Assess vulnerability
Mangrove distribution	Vector	Giri et al., 2011	Mangrove cover	Assess vulnerability
Seagrass distribution	Vector	UNEP-WCMC and Short FT 2018	Seagrass cover	Assess vulnerability
Coral Triangle	Vector	Cros et al., 2014	Boundary	Assess vulnerability

map) to ensure the completeness of the dataset. We increased the accuracy of Reed and Trubetskoy (2019)'s spatial dataset for existing routes by identifying and replacing road and rail polylines using the more spatially accurate Open Street Map (OSM) spatial data (https://wiki.openstreetmap.org/wiki/Shapefiles). Marine routes were digitised according to the digital map extracted from the National Administration of Surveying, Mapping, and Geoinformation of China. The routes were assessed and validated for a scale of 1:250,000, which is the equivalent of a spatial resolution of around 250 m, though in most cases the resolution was much higher. While the OSM data has a positional accuracy of at least 1:5000.

Our study area is divided into two parts: the terrestrial and marine realms. The terrestrial analyses focused on mainland Southeast Asia (Singapore, Peninsular Malaysia, Thailand, Vietnam, Laos, Cambodia, and Myanmar). Whereas the marine analyses are based off both continental and insular Southeast Asia (Indonesia, Timor-Leste, Philippines, Brunei, and Malaysian Borneo; Fig. S1).

To assess the potential impacts of BRI's land and marine routes on biodiversity, we used 13 spatial datasets representing terrestrial and marine biodiversity and also protection status and condition (Table 2). These datasets included Protected Areas (PA), Key Biodiversity Areas (KBA), terrestrial ecoregions, forest cover, threatened species (mammals, birds, amphibians, and reptiles), the human footprint, as well as fragile coastal ecosystems (coral, seagrass, and mangrove distribution; Table 2) and the boundary of Southeast Asia's coral reef triangle. All spatial data were harmonized to the same extent based on the Southeast Asian administrative boundary and are fit for use at spatial resolutions >1 km pixel size and for estimating area of overlap.

The dataset on PAs was obtained from the World Database on Protected Areas (WDPA), August 2019 version. We used a conservative analysis which only considered PAs that were characterised by polygons (the database includes PAs characterised by a point location). We identified 2020 PAs in Southeast Asian countries. Following previous global assessments (e.g. Watson et al., 2014), we removed PAs that are "proposed" and designated as "international", including only PAs that are designated at the national level. This resulted in 1867 PAs, including those with no IUCN classification. In 509 of these PAs, we found some spatial overlap between the boundaries as PAs with different designations can overlap (mean area of overlap ± standard deviation = 692 km^2 ± 2003). We followed WDPA best practice guidelines on calculating national area coverage (https://www.protectedplanet.net/c/calculating-protected-area-coverage) and combined polygons that overlapped into single units, then calculated the number and area at a country level. For the marine analyses we only included marine PAs of categories "1" (i.e. Coastal) and "2" (i.e. 100% marine). Overall, we assessed 472 terrestrial and 678 marine PAs.

A revised version of Olson's terrestrial ecoregions of the world map developed by Dinerstein et al. (2017) (https://ecoregions2017.appspot.com/) was used to characterise ecoregion boundaries and their vulnerability. Based on an ecoregions' extent of remaining natural habitat and protected land, Dinerstein et al. (2017), categorised ecoregions assigned to Nature Needs Half (NNH) categories. Mainland Southeast Asia includes 38 ecoregions.

We used the Global Human Footprint map of 2009 developed by Venter et al. (2016) to characterise level of modification (i.e. natural versus modified landscapes). The Human footprint (1-km pixel size) was classified to five levels of increasing intensity of human modification on the landscape: 1–10, 11–20, 21–30, 31–40, and 41–50 (Venter et al., 2016). Forest cover of Southeast Asia was obtained from Li et al. (2016) who identified predominantly "natural forest" from other woody land covers such as oil palm and rubber which are often misclassified as forest.

For the analysis of threatened species, we used distribution maps for birds, mammals and amphibians from IUCN, and reptiles from the Global Assessment of Reptile Distributions (GARD) to identify areas of high concentrations of threatened fauna that may be impacted by BRI. For all four taxonomic groups, we focused on threatened species, which are those listed by the IUCN Red List as Critically Endangered (CR), Endangered (EN) or Vulnerable (VU). In the analysis we included all extant native species and excluded species classified as 'Extinct', 'Introduced' or 'Vagrant' within our study area (Jenkins et al., 2013). This resulted in a total of 371 species (birds = 94, mammals = 101, amphibians = 79, and reptiles = 97). We then generated species richness maps by overlaying the distribution range of species within each of the four broad taxonomic groups. This methodology is commonly used (e.g. WWF, 2017; Di Minin et al., 2019), but since many species, especially among amphibians, are data-deficient or have not been included in IUCN range analysis, the results must be interpreted cautiously (Hughes, 2017a, 2017b).

KBAs obtained from were clipped to mainland Southeast Asia to include only terrestrial KBAs. To delineate KBAs in marine areas, we clipped the layer according to the Maritime Exclusive Economic Zone (EEZ) Boundaries (Flanders Marine Institute, 2018). A total of 458 terrestrial and 260 marine KBAs were analysed. The KBA data also included an assessment of their protection status which we used in the analysis. While Coral reef, Mangrove and Seagrass distribution data were obtained from a range of sources.

2.2. Impact zones and threshold distances

We examined scale-dependent impacts of terrestrial infrastructure and marine routes associated with the proposed BRI routes using four overlapping buffer distances considered ecologically significant to represent 'impact zones'. Specifically, impact zones of <1, <5, <25, and <50 km were assessed along either side of all marine and terrestrial routes rail and road infrastructure development types. The impact zone buffer distances were based on a literature review of the direct and indirect impacts of linear infrastructure and follow similar studies using buffer zones to assess potential biodiversity impacts due to BRI from linear infrastructure (Hughes, 2019; Hughes et al., 2020; WWF, 2017) and ports (Turschwell et al., 2020).

The application of buffer zones is a useful way to quantify negative effects of linear infrastructure. It allows regional planners to estimate the potential extent of impact of an existing or proposed infrastructure. The extent of road's ecological effects into adjacent landscapes is well documented (e.g. the so-called 'road-effect zone' or 'impact zone'; Alkemade et al., 2009; Forman, 2000; van der Ree et al., 2015) but less is known for railways and marine routes. The ecological mechanism, likelihood, magnitude, and timescale of these impacts will vary with distance and infrastructure type. For road and rail there are four general effects: i.e., mortality (commonly from collision), barrier effects, disturbance (e.g., noise), and habitat loss and fragmentation (direct and indirect) (Barrientos et al., 2019; van der Ree et al., 2015).

Ecological impacts at distances up to 5 km commonly represent direct impacts of linear infrastructure and the distances at which impacts are likely to be the greatest (e.g. Benítez-López et al., 2010). The threshold distance of 1 km represents the highest level and variety of road impacts, indicative of biotic effects, such as depressed species abundance and diversity, noise aversion and forest edge effects, in addition to easy access to various forest resources and heightened hunting probabilities. A study based in the Congo Basin showed that 76% percent of agricultural clearings were within 1 km of a public road (Cordero-Sancho and Bergen, 2017). Another study cited forest loss of up to 900 m of roads combined with forest-edge losses (Sloan et al., 2014).

The next threshold we applied is a 5-km impact zone. Studies in Southeast Asia show that the majority of deforestation occurs within 2.5 km of roads (Hughes, 2018). Benítez-López et al. (2010) cited population density declines of bird and mammal extending up to 1 km and 5 km respectively, with observed infrastructure avoidance in open areas as opposed to forested areas. A systemic review by (Clements et al., 2014) revealed in Peninsular Malaysia, ~90% of snares and poaching camps were located within 5 km of a paved road. At larger spatial scales of 5–25 km from roads, hunting of wildlife creates zones of elevated mortality and animal avoidance, and could extend much further for wide-ranging species such as elephants (Blake et al., 2007).

The 25 and 50 km impact zones reflect broader environmental changes along infrastructure (Laurance et al., 2015b). In the case of roads, this is usually characterised by subtle, unanticipated, and/or difficult-to-observe secondary effects (Sloan et al., 2017), for instance, spontaneous agricultural conversions in the Amazon forest (Barber et al., 2014), illegal logging in Southeast Asia of up to ~20–30 km (Linkie et al., 2014), and poaching in Africa (~80 km) (Blake et al., 2008). The 50 km impact zone (25 km to 50 km buffer distances) represents the maximum impact distance in our analyses, which includes most of the intense impacts and developments linked to the road. Additional supportive infrastructure to generate energy and provide raw materials to build the road may be sourced from a much larger area but cannot be easily estimated (Hughes, 2019). In Amazonia, paved and unpaved roads can induce forest clearings of up to 18–45 km (Southworth et al., 2011), while deforestation spillover are reported up to 100 km to larger roads (Pfaff et al., 2007).

Compared to roads, less is known about the ecological impacts of railways (Barrientos et al., 2019; Dorsey et al., 2015; Popp and Boyle, 2017). Although the specific impacts of road and rail may differ, the types of impacts are similar (e.g. direct mortality, barrier effects, disturbance and fragmentation) and likely to occur at similar distances (see Barrientos et al., 2019; Dorsey et al., 2015; Popp and Boyle, 2017 for reviews of rail impacts which include comparisons with road impacts). Mortality due to wildlife train collisions is well documented, but rail involves other forms of mortality from electrocution and entrapment (Barrientos et al., 2019; Dorsey et al., 2015). Low traffic volumes associated with rail infrastructure intuitively would indicate that rail mortality is lower, however, periods of time without traffic may actually mean that animals attempt to cross and then become trapped on the tracks (Popp and Boyle, 2017). Rail can be both a physical and behavioural barrier (Borda-de-Água et al., 2017), especially high speed rail which often includes fencing or steep ditches. The evidence for impacts of fragmentation from rails is lacking, however Barrientos et al. (2019) suggested that the fragmentation effects attributed to roads may also apply to railways. In some cases, the impacts of rail may be greater than those occurring due to roads (Dorsey et al., 2015). For example, Huber et al. (1998) found brown bear mortality in Croatia associated with rail to be was equal to or higher than roadkill rates.

A further, complicating factor associated with estimating impacts from rail in comparison to roads, is that many train lines commonly include maintenance and emergencies access roads (Losos et al., 2019), as well as roads built to support construction and thus may include a mix of road impacts. Using before and after satellite imagery we found that the Kunming-Vientiane high speed rail, being built under the BRI primarily in intact northern Laos tropical forest landscapes, appears to be associated with spillover effects (i.e. indirect fragmentation due to land cover conversion) and usually includes access roads to and from the rail, as well as alongside the rail line (Fig. S2).

For the terrestrial routes we used five categories of linear transport infrastructure adapted from Reed and Trubetskoy (2019) to characterise routes based on potential ecological impacts: new road, new high capacity rail, new rail, road upgrade and upgrade to high capacity rail (see Table S1 for examples of each). New roads and rail are likely to have major impacts due to increased accessibility in frontier landscapes. It must be noted however that the impacts of high speed rail are likely to be different because it has few stations and thus does not open new landscapes in the same way, though may still be followed by new roads for maintenance and local use. We didn't include specific categories for upgrade to high speed rail (i.e. fast trains) versus upgrade to high capacity rail (i.e. more tracks) as in most cases an upgrade would preserve the existing rail infrastructure and/or the design of new train infrastructure described by publicly available documents was unclear with train speeds ranging greatly, even within a single project. For example, it has been reported that the Hanoi-Ho Chi Minh City rail project will include trains travelling at 50–60 km/h and 350 km/h (See Table S1 for further description of each route based on publicly available information and from Reed and Trubetskoy (2019)).

For interpreting our analysis, buffer distances close to BRI routes, new infrastructure and roads rather than rail are likely to have greater impacts. Thus, considering only impacts at 5 km from BRI routes represents an optimistic scenario, while 50 km represent a pessimistic projection of impacts including secondary impacts such as development, supportive infrastructure and various extractive activities such as logging, agriculture, hunting. While it is likely that ecological impacts of rail from <1 km and <5 km and even <25 km are similar to roads, it is questionable whether the spillover effect associated with increasing accessibility will result in land conversion at distances of 25 km to 50 km (but see Fig. S2). Given the uncertainty, in this paper we report these distances in the plots but for rail we mark them with dotted lines to indicate that these distances are highly speculative. Also, within the text we have focused on the distances <25 km to any kind of infrastructure.

In the marine environment, we considered shipping routes as analogous to terrestrial road systems as they facilitate transportation, connect locations, and concentrate vessel movements (Pirotta et al., 2019). However, there is very little guidance on impact zones in marine areas (Di Minin et al., 2019). Thus, we retained the impact zones of 1, 5, 25, and 50 km to allow for comparison between terrestrial and marine routes (Jones et al., 2018), although we recognize that the maritime routes are coarse. In addition, the comparison of buffer distances can be used to demonstrate whether route location has a great influence on the results, i.e., if changing buffer sizes changes the results dramatically, the accuracy of route locations is then critical for interpreting the outputs. While a similar result at various distances indicates that biodiversity is homogenous across the study area.

2.3. Spatial analysis

We examined the impacts of BRI routes' by overlaying their impact zones onto the terrestrial ecosystems, terrestrial biodiversity, and marine ecosystem indicators. All analyses were made using the World Cylindrical Equal Area projection, which minimises distortions associated with area-based calculations.

We quantified the proportion of area of each impact zone which overlapped with threatened species, PAs, KBAs, and terrestrial ecoregions. Similarly, we assessed the extent of impacts of BRI marine route on marine PAs, KBAs, and fragile ecosystems – corals, mangrove and seagrass. PAs were also intersected with the BRI routes to analyse the number of PAs directly bisected by the BRI routes. Overlapping buffers were used for all analyses (i.e. 0 to 1 km; 0 to 5 km; 0 to 25 km and 0 to 50 km) apart from the assessment using forest cover and the human footprint. For forest cover and the human footprint, the measurements in the buffer areas were mutually exclusive, as we were interested in whether the BRI routes were found in already developed/impacted areas. Here, we quantified the proportion of area within 0 to 1 km, 2 to 5 km, and subsequently at 5-km intervals up to 50 km.

3. Results

3.1. BRI impacts on terrestrial ecosystems

The proposed infrastructure routes intersect protected and key biodiversity areas, important ecosystems, forests, and wilderness areas (Fig. 1). A total of 21 PAs (4% of 472 PAs) across mainland Southeast Asia are directly bisected by the BRI routes which traverse through 210 km of protected habitat (See Table S2). Some 50% of the bisected PAs are of those managed exclusively for biodiversity conservation (IUCN classes I-IV; Table S2). In Cambodia a new road directly bisects one protected area, while projects involving road upgrades bisect a total of four PAs across Laos (n = 2), Vietnam (n = 1) and Myanmar (n = 1). The development of new rails poses the greatest threats as it bisects eight PAs (including six national parks), the highest number of PAs bisected among all route types. BRI projects involving rail upgrades to high capacity rail occur within five PAs in Vietnam and one in Thailand. Lastly, newly built high capacity rail will bisect one PA from Myanmar and one in Malaysia (Table S2). The PAs with the longest distance bisected by BRI routes are the Nam Kan National Park in Laos (45 km), Hai Van-Hon Son Tra marine protected area in Vietnam (34 km), and Khao Laem National Park in Thailand (28 km) (Table S2).

New roads overlap with two PAs within the 1 km buffer zone and with five PAs at 5 km. Within the 25 km buffer, new roads intersect with seven PAs, with an overlapping area of 1684 km^2, covering 16% of the 25 km impact zone area (Fig. 2a). Among these PAs, Cambodia has the highest number of PAs overlapped (n = 6), followed by Vietnam (n = 1). Road upgrade may affect up to 13 PAs within 25 km, with Laos having the greatest proportion of affected PA area (at 3131 km^2, 6% of 25 km impact zone area). While, Vietnam (n = 6) has the highest number of PAs intersected at 25 km, followed by Myanmar (n = 5), Laos (n = 2) and Thailand (n = 1) (Fig. 2b). There are upto 13 PAs within 1 km of new rails amounting to an area of 133 km^2 (Fig. 2c), and 32 PAs within 25 km (9% of 25 km impact zone). PAs in Thailand account for the greatest proportion area (6%) of 25 km of railways. The upgrade from standard rail to high capacity rail has the highest overlap in terms of number of PAs at 25 km, representing 17% (n = 80) of PAs in mainland Southeast Asia (Fig. 2d). There are 15 PAs within 25 km of new high capacity rail, covering 1.6% proportion area (Fig. 2e). Overall, across all BRI route types 27% (n = 126) of PAs in mainland Southeast Asia fall within 25 km of routes, with Vietnam (n = 46) having the highest numbers of overlapped PAs, followed by Thailand (n = 35), Malaysia (n = 21), Cambodia (n = 8), Myanmar (n = 7), Laos (n = 5) and Singapore (n = 4).

Fig. 3 shows that the areal extent of overlap with KBAs is greater than PAs. At the 25 km impact zone, 20% of new roads overlap with 10 KBAs in Cambodia (Fig. 3a), whereas 13% of road upgrade overlap with 22 KBAs, of which Laos covers the highest proportion area (6% of 25 km impact zone), while 64% (n = 14) of affected KBAs are in Myanmar (Fig. 3b). Up to 14 KBAs are within 5 km of new rails, of which Thailand occupies the largest proportion area (4.6% of impact zone) (Fig. 3c), whereas up to 31 KBAs (7.9% of impact zone) is within 5 km of rail upgrade to high capacity rails (Fig. 3d). New high capacity rails on the other hand overlap with up to 11 KBAs at the 5 km impact zone (4.3% of impact zone area) (Fig. 3e). Overall, at 5 km distance, a total of 60 KBAs overlap with the BRI infrastructure routes across all route types while up to 28% (n = 129) of 458 terrestrial KBAs are within 25 km of all BRI routes. Of 129 KBAs which are within 25 km of the BRI, 55% (n = 71) have little, no, or unknown formal protection status, or are not listed as legally protected. Country-based patterns varies from that of PAs, with Vietnam (n = 36) having the highest number of KBAs impacted at the 25 km impact zone, followed by Thailand (n = 31), Myanmar (n = 27), Malaysia (n = 14), Cambodia (n = 11), Laos (n = 7), and lastly Singapore (n = 3).

Ecoregions affected along the BRI vary in proportion area across each impact zones based on route type (Fig. 4). New roads overlap with ecoregions listed as Nature Imperilled and Nature Could Reach Half Protected, of which the proportion area of Nature Could Reach Half Protected ecoregions covers 68% of area within 1 km of new roads. However, the proportion area of Nature Could Reach Half Protected ecoregions (less than half is protected but there is enough remaining to reach Half Protected) decreases further away from the route, as the area of Nature Imperilled ecoregions (the amount of natural habitat remaining is less than or equal to 20%) increases. Regions where road upgrades occur overlap predominantly with Nature Imperilled and Nature Could Reach Half Protected ecoregions. The proportion area of ecoregions within new rails exhibit similar trends as that of new roads and road upgrades. Nature Could Reach Half Protected ecoregions overlap with 56% of 1 km impact zone area from new rails and decreases to 54% at 5 km away as that of Nature Imperilled and Nature Could Recover ecoregions increases.

Overall, a total of 31 ecoregions are within 5 km of the BRI across all route types, of which 39% (n = 12) are listed as Nature Imperilled, 42% (n = 13) are listed as Nature Could Reach Half Protected and 19% (n = 6) are listed as Nature Could Recover (Table S3). Furthermore, 16% (n = 5) of the 31 ecoregions have >50% of their total area within the 25 km impact zone. These include the Nature Imperilled Chao Phraya lowland moist deciduous forests (50% or Area = 10,188 km^2) and the Red River freshwater swamp forests (63% or Area = 10,724 km^2). The Northern Vietnam lowland rain forests and Southern Vietnam lowland dry forests are among the ecoregions that have >70% of their areas overlap within 25 km of BRI routes.

Figs. 5 and 6 show the proportion of human footprint classes and forest cover for a range of distances for each BRI route type. Here, we quantified the proportion area of human footprint classes and forest cover within 0 to 1 km, 2 to 5 km, and subsequently at 5-km intervals up to 50 km. In these analyses (Figs. 5 and 6) the buffer areas were mutually exclusive, as we were interested in whether the BRI routes were found in already developed/impacted areas. Across all route types, the proportion area of impact zones occupied by Human

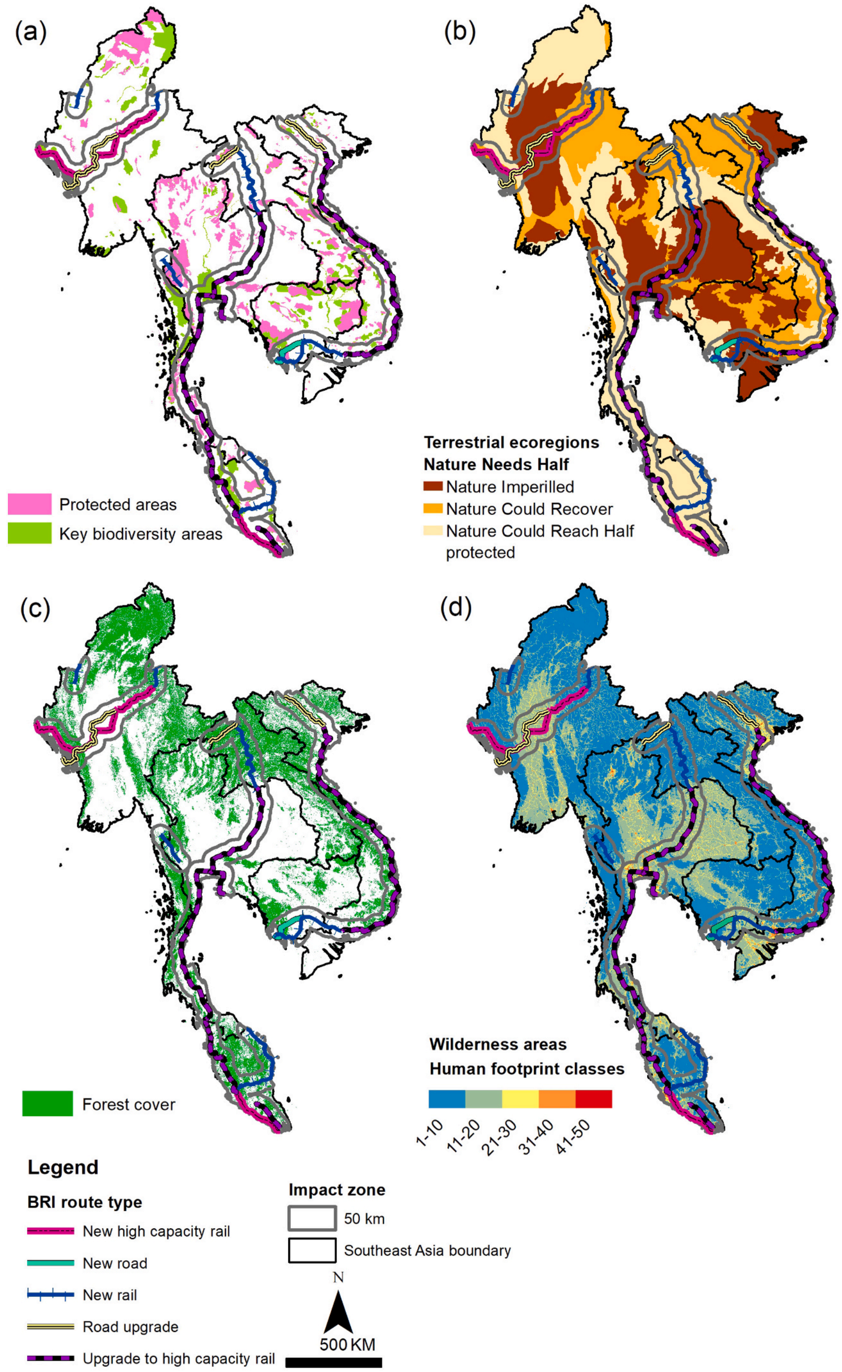

Fig. 1. Map of Belt and Road (BRI) routes in mainland Southeast Asia based on ecologically sensitive areas: (a) Protected areas and key biodiversity areas, (b) Terrestrial ecoregions of the world, (c) Forest cover, and (d) Wilderness areas. The BRI land route indicates the presence of potential infrastructure in the region, while the extent of negative influence of BRI on adjacent land is represented by a 50 km buffer along the routes known as "impact zones.

Footprint Class 1–10 (which characterises more natural areas), is the lowest near BRI routes and increases with distance from the BRI routes (Fig. 5). This is followed by a plateauing of the increase in Human Footprint Class 1–10 at distances around 11–15 km or 16–20 km i.e. areas close to BRI routes are more disturbed. Conversely, the proportion area of impact zones occupied by human-modified classes is the highest nearest the BRI routes and decreases further away from the BRI routes. Projects involving upgrade to high capacity rails generally occur on already developed lands, where Class 1–10 only covers 1.2% of area between 0 and 1 km, amounting to an area of 104 km^2. A similar pattern to the human footprint classes was found with forest cover. The proportion area of impact zones overlapped with forest cover is lowest near the BRI routes and increases further away from the BRI (Fig. 6).

3.2. BRI impacts on terrestrial biodiversity

Fig. 7a–e shows the percentages of threatened taxa groups with ranges impacted across various impact zones along the BRI routes types. The number of species which overlapped with the BRI routes varied with route type and distance and taxon. Rail had higher percentages of threatened species overlapping the BRI impact zones compared to roads. For example, the highest impacts to threatened amphibians within the 1 km impact zone was from new rails (6.3% or n = 5) (Fig. 7c), while at 5 km the highest impacts to threatened amphibians

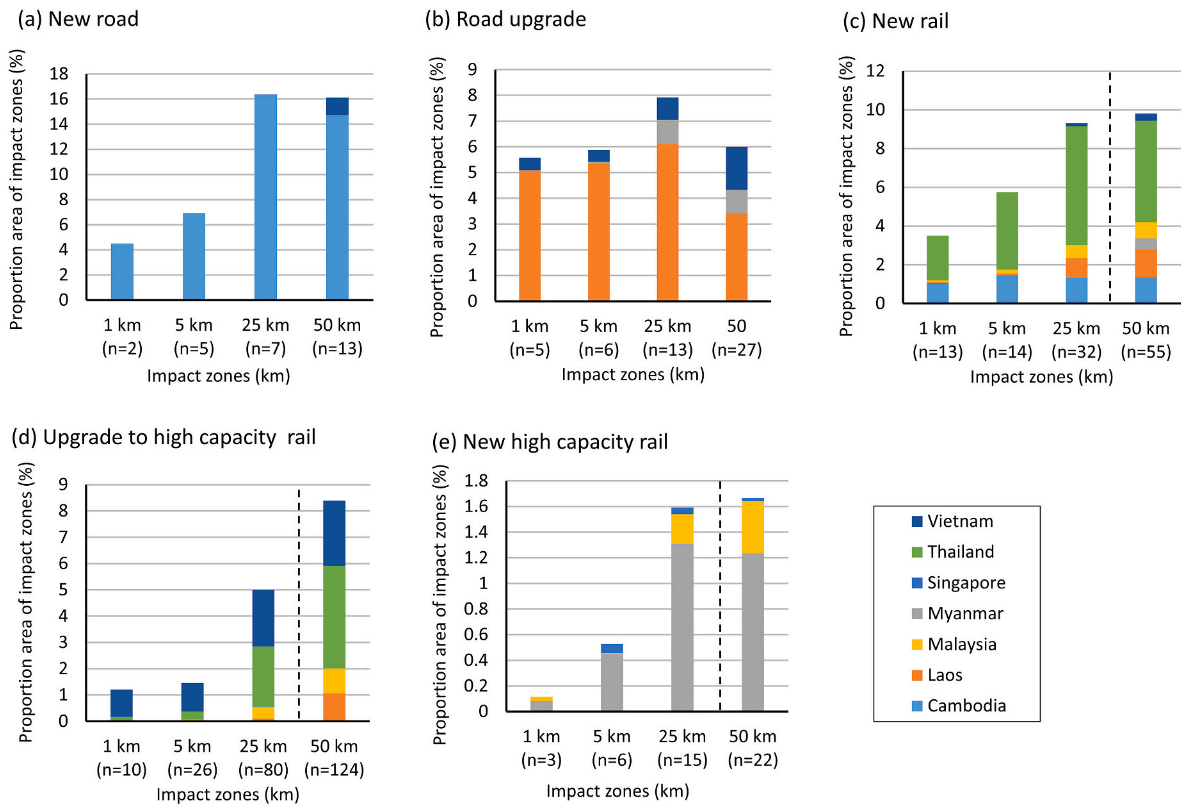

Fig. 2. Proportion area of impact zones that overlap with PAs across the types of BRI routes: (a) New road, (b) Road upgrad,e (c) New rail, (d) Upgrade to high capacity rail, and (e) New high capacity rail. Total number of PAs within 1 to 50 km of the BRI routes in brackets on x-axis. Note that impacts >25 km for rail are speculative, as indicated by the dashed line.

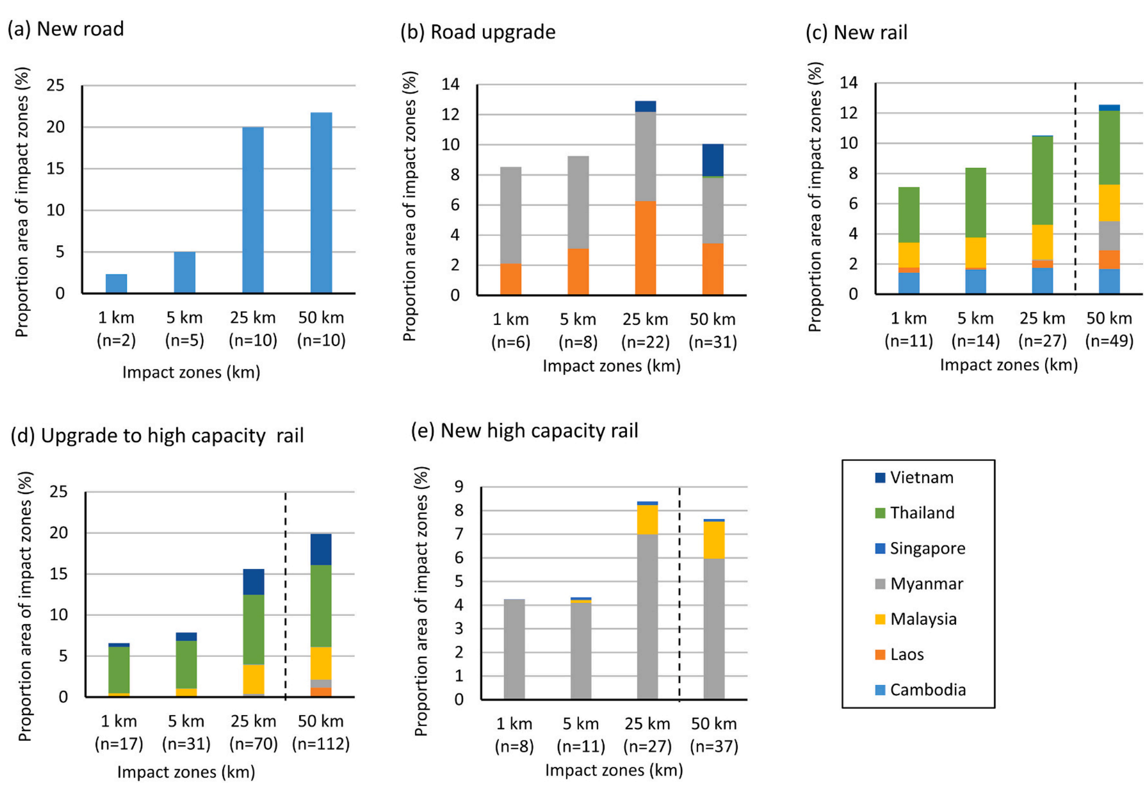

Fig. 3. Proportion area of impact zones that overlap with KBAs across the types of BRI routes: (a) New road (b) Road upgrade (c), New rail (d), Upgrade to high capacity rail, and (e) New high capacity rail. Total number of KBAs within 1 to 50 km of the BRI routes in brackets on x-axis. Note that impacts >25 km for rail are speculative, as indicated by the dashed line.

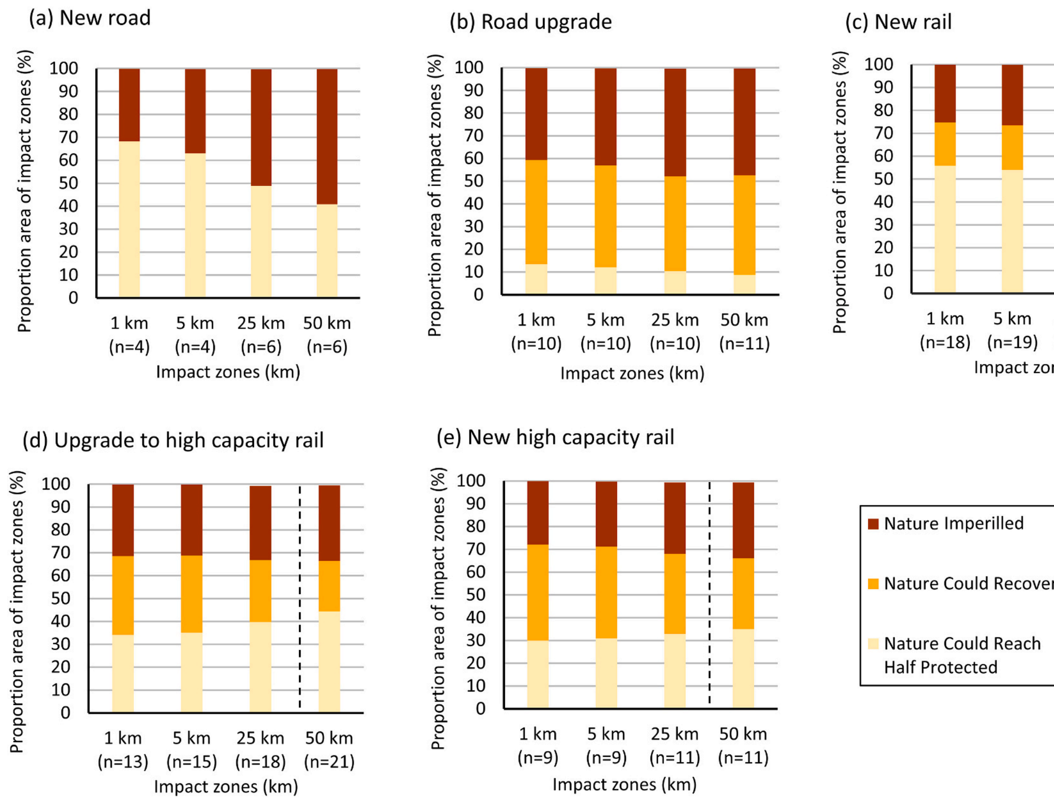

Fig. 4. Proportion area of impact zones overlapped with terrestrial ecoregions by Nature Needs Half status across the types of BRI routes: (a) New road, (b) Road upgrade, (c) New rail, (d) Upgrade to high capacity rail, and (e) new high capacity rail. Total number of ecoregions affected in brackets on x-axis. Half Protected (>50% of total ecoregion is protected), Nature Could Reach Half (<50% total ecoregion area is protected but sum of total ecoregion and unprotected natural habitat is >50%), Nature Could Recover (the sum of the amount of natural habitat remaining and the amount of the total ecoregion that is protected is <50% but >20), and Nature Imperilled (the sum of the amount of natural habitat remaining and the amount of the total ecoregion that is protected is less than or equal to 20%). Note that impacts >25 km for rail are speculative, as indicated by the dashed line.

was due to new rail and upgrade to high capacity rails (7.6% or n = 6) (Fig. 7d). In contrast for new roads, the number of threatened birds remain constant across all distances (20% or n = 9). In the BRI's 25 km impact zone, the highest number of threatened mammals was in Vietnam and Peninsular Malaysia (Figs. 8a, S3e) and high concentrations of threatened birds were found in Myanmar and Peninsular Malaysia (Figs. 8b, S3f). The distribution of threatened amphibians appeared patchy, with the highest richness in small regions in Vietnam (Figs. 8c, S3g). In addition, the highest richness of threatened reptiles was in Vietnam (Figs. 8d, S3h). Overall, 83% (n = 82) of threatened mammals, 78% (n = 73) of threatened birds, 53% (n = 42) of threatened amphibians and 52% (n = 50) threatened reptiles have ranges within 25 km of BRI routes.

Within 5 km of all the route types, 15% of threatened mammals (n = 12) have ranges affected between 10%–23% of their total area. Within 25 km for all route types, the critically endangered large-antlered muntjac (*Muntiacus vuquangensis*), endangered Malayan tapir (*Tapirus indicus*), white-handed gibbon (*Hylobates lar*), and vulnerable Sumatran Serow (*Capricornis sumatraensis*) are among the large charismatic mammals that have 10–28% of their ranges potentially impacted (Table S4). For threatened birds, 23% (n = 15) have ranges affected between 10%–20% of their total area within 5 km, while 10% (n = 7) have >50% of ranges intersected by the 25 km impact zone, this includes the critically endangered *Lophura edwardsi* which has 81% of its range affected (Table S5). For threatened amphibians, the endangered *Ingerophrynus gollum* (21% of range) and *Kalophrynus palmatissimus* (32% of range) are the impacted the greatest within 5 km. Moreover, 19% (n = 8) have >50% of ranges intersected within 25 km, of which two are endangered, two vulnerable, and one critically endangered species have 100% of their ranges within the 25 km impact zone (Table S6). Lastly, two species of thereatened reptiles have >50% of ranges impacted within 5 km of all route types. These include the critically endangered *Cyrtodactylus guakanthanensis* and vulnerable *Gekko russelltraini*, whereas 40% (n = 20) of threatened reptiles have >50% of ranges within the 25 km of the BRI, among these, 11 species have their entire range within the 25 km impact zone (Table S7). Additional species not included in IUCN range maps may also be effective but could not be assessed due to the lack of data.

3.3. BRI impacts on marine ecosystems

Fig. 9 shows the BRI's marine route and 50 km impact zone overlapping with ecologically sensitive areas across insular Southeast Asia. A total of 20 marine PAs and 16 marine KBAs in insular Southeast Asia are potentially affected within 50 km of BRI's marine route (Fig. 10a–b). Indonesia is the most impacted country, with a total of 12 marine PAs, amounting to an area of 17,887 km^2 or 1.76% of the 50 km impact zone area, followed by Vietnam with 7 marine PAs affected (Area = 1260 km^2 or 0.12%), and lastly Malaysia with only 1 marine PA affected (Area = 0.54 km^2 or 0.0005%).

For key marine ecosystems (Fig. 10c–e), a total of 1026 km^2 of mangroves are within the 50 km impact zone, with Indonesia being the most impacted country (Area = 811 km^2 or 0.08%), followed by Malaysia (Area = 187 km^2 or 0.02%) and then Vietnam (Area = 35 km^2 or 0.003%). While for seagrass habitats 439 km^2 are within the 50 km impact zone with 280 km^2 in Indonesia (0.27%) and 159 km^2 in Malaysia (0.015%). Lastly, 2780 km^2 of Southeast Asian reefs are potentially affected within the 50 km impact zone with Indonesia having the highest reef area overlap (Area = 2734 km^2 or 0.26%) followed by Vietnam (Area = 45 km^2 or 0.004%).

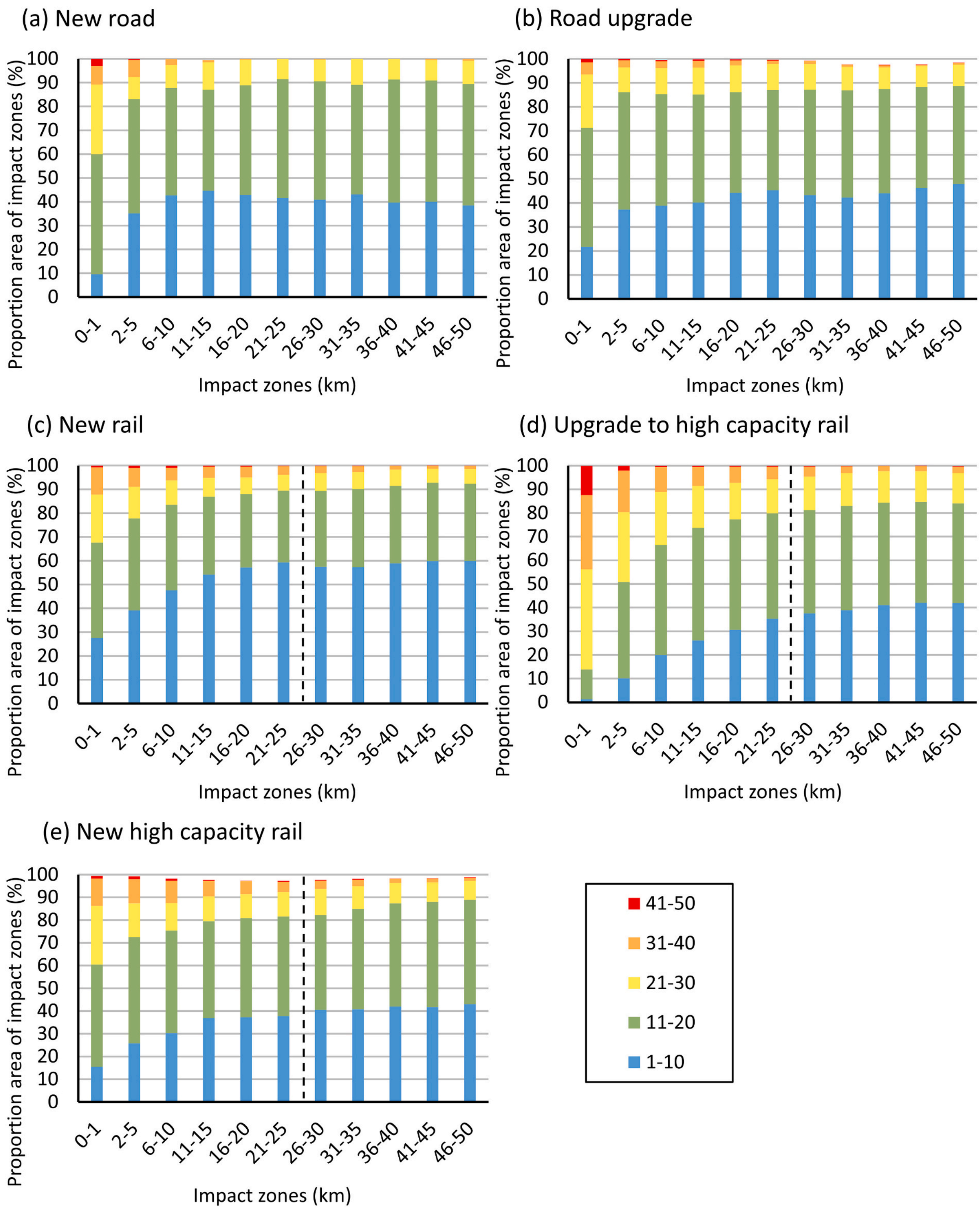

Fig. 5. Proportion area of impact zones that overlap with human footprint areas across non-cumulative impact zones of 1 km, 5 km, and subsequently 5-km intervals up to 50 km across the types of BRI routes: (a) New road, (b) Road upgrade, (c) New rail, (d) Upgrade to high capacity rail, and (e) New high capacity rail. Human footprint is denoted by human footprint classes which represent the intensity of human influence on the landscape (i.e. Class 1–10 = wilderness areas; Class 11–20 to Class 41–50 = increasing human modified/developed areas). Note that impacts >25 km for rail are speculative, as indicated by the dashed line.

4. Discussion

4.1. Does BRI traverse undisturbed terrestrial areas?

We found that areas closer to BRI routes generally were less intact (i.e. urban and agricultural areas or highly disturbed ecosystems), with less forest cover, and relatively high human footprint values than areas further from BRI routes. We also found distances of 0 to 10 km from new rail had lower human footprint values compared to other route types and thus are likely to be built in more intact landscapes, which follows the common assumption and concerns that new infrastructure are more likely to be found in intact or less disturbed landscapes. In

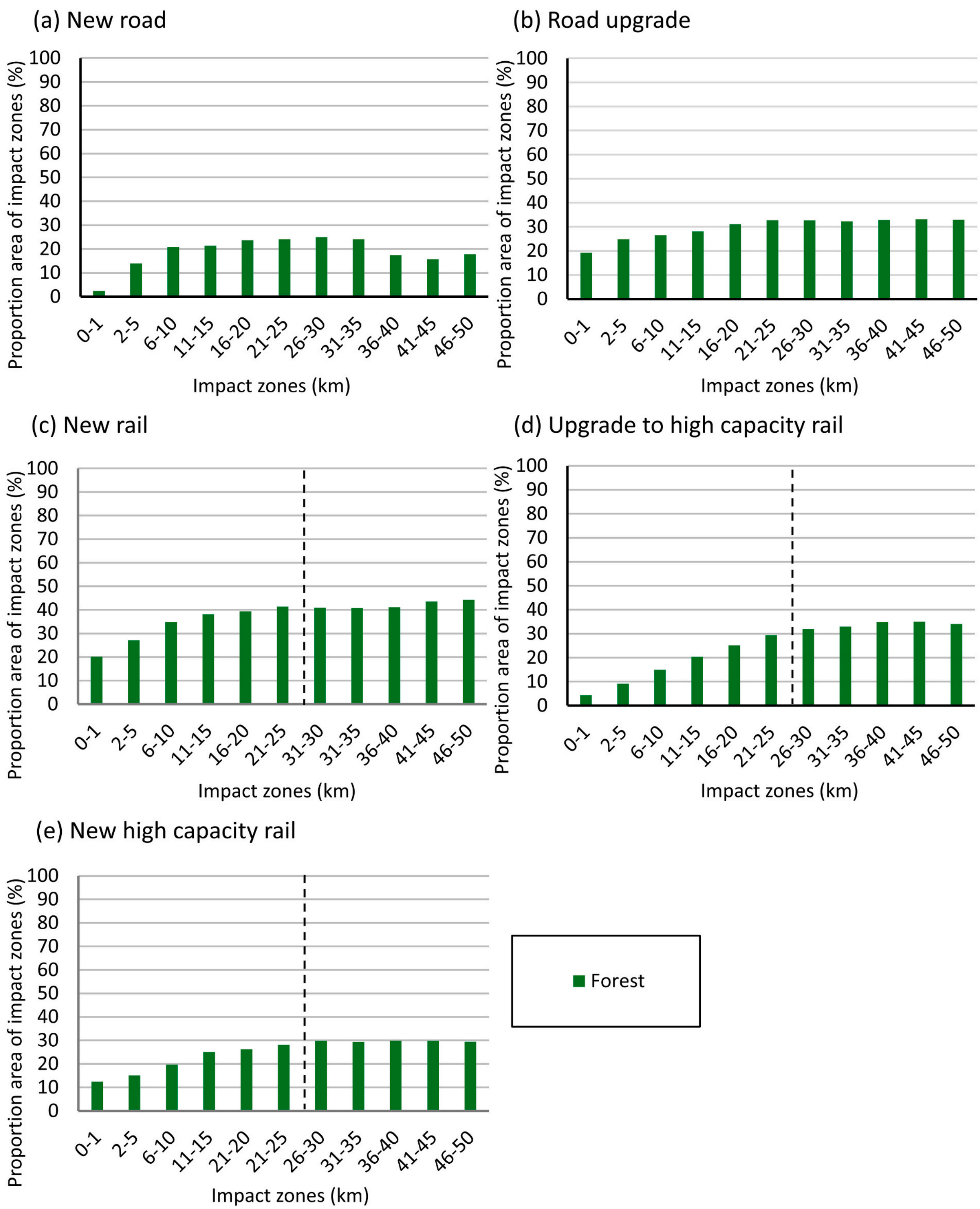

Fig. 6. Proportion area of impact zones that overlap with forested areas across non-cumulative impact zones of 1 km, 5 km and subsequently 5-km intervals up to 50 km across the types of BRI routes: (a) New road, (b) Road upgrade (c), New rail, (d) Upgrade to high capacity rail, and (e) New high capacity rail. Note that impacts >25 km for rail are speculative, as indicated by the dashed line.

contrast new road did not exhibit such patterns, for example at 2–5 km from the road only around 14% is composed of forest cover (Fig. 6), though this may be because new road routes are represented by a single 226 km long project in Cambodia. Road upgrades were generally found within already disturbed landscapes and the levels of disturbance remained almost constant with distance. By length, new rail represents 21% of project infrastructure development type compared to upgrade to high capacity rail with a total length of 4475 km, comprising 49% of the total BRI length within our study area. At coarse scales our analysis suggest that new rail is perhaps the most concerning for biodiversity, as it was commonly found in more intact landscapes (Figs. 5 and 6). From the ecological perspective, new rail has the potential for opening up new landscapes, though this will depend on a number of factors such as how many stations are built, if routes are elevated or at ground level

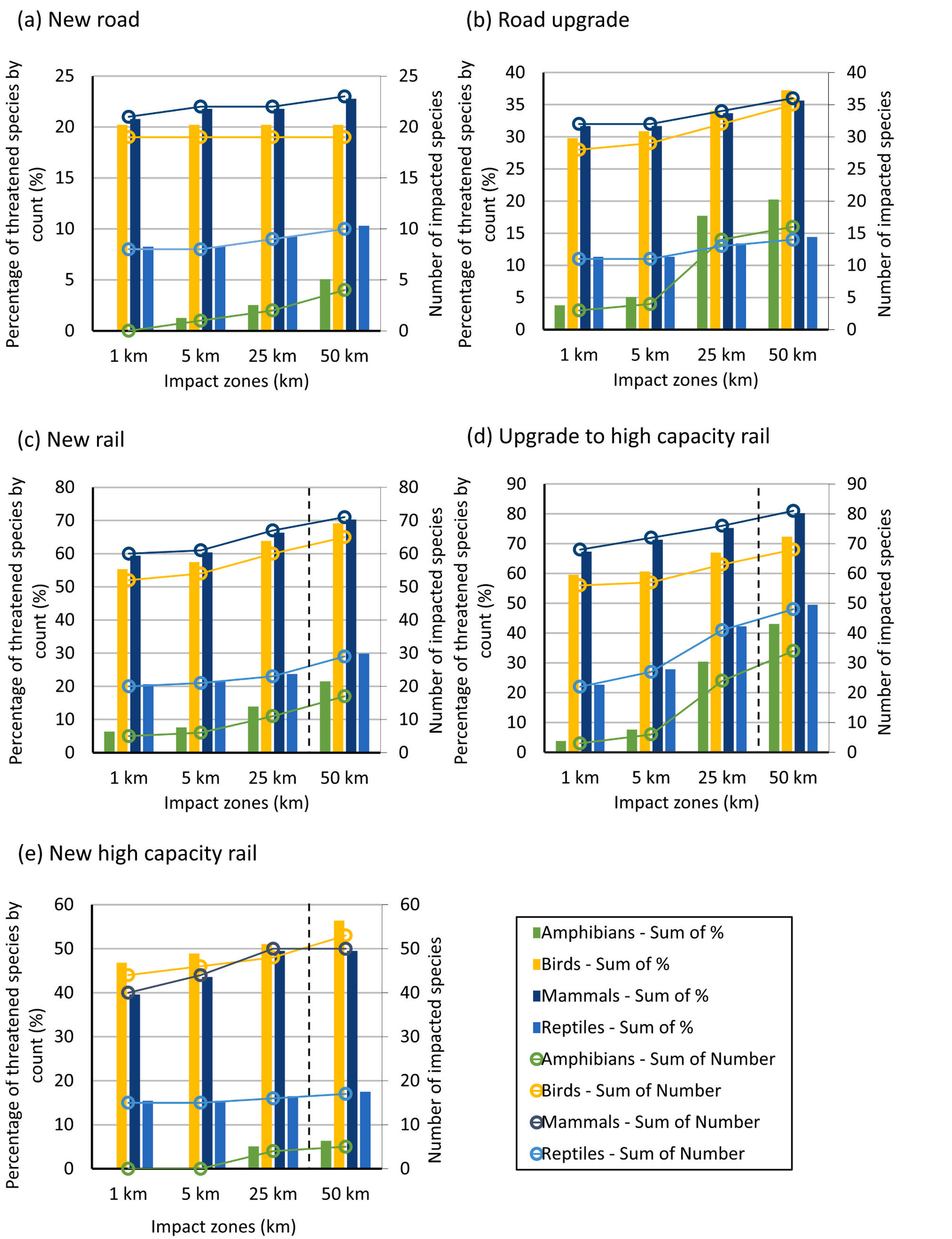

Fig. 7. Percentage of the total number of species (primary y-axis) and number (secondary y-axis) of species from threatened taxa groups in mainland Southeast Asia with distributive ranges overlapped within the impact zones across BRI route types: (a) New road, (b) Road upgrade, (c) New rail, (d) Upgrade to high capacity rail, and (e) New high capacity rail. Note that impacts >25 km for rail are speculative, as indicated by the dashed line.

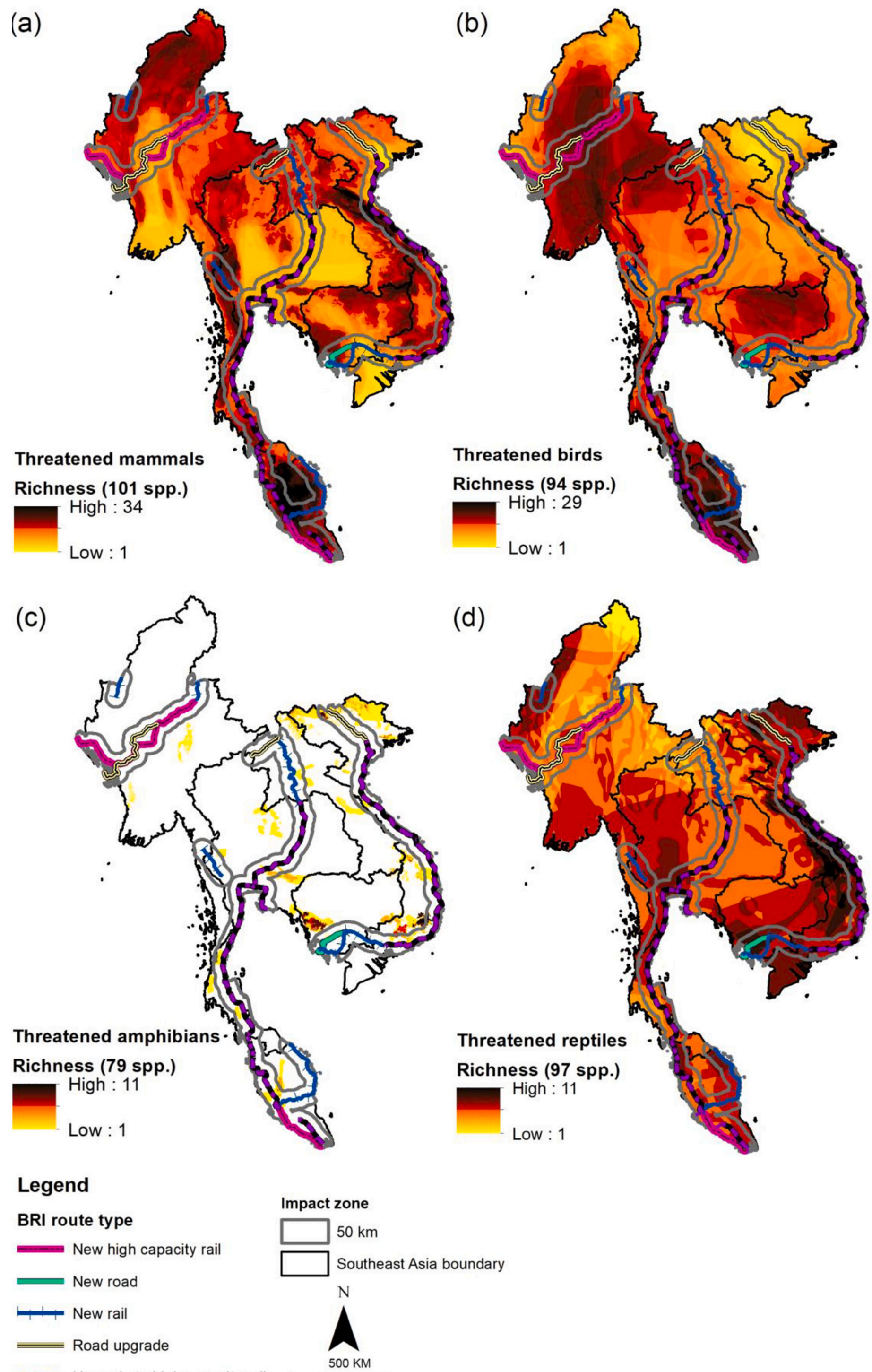

Fig. 8. Threatened species richness of (a) Mammals (b) Birds (c) Amphibians and (d) Reptiles in mainland Southeast Asia within the 50 km impact zone across the types of BRI routes. Richness indicates the number of overlapping species within a map unit of 250 m.

and also the existence of supporting infrastructure.

Some of Southeast Asia's last intact forests are already reported to be under threat of either existing or proposed linear infrastructures such as roads and railways (McCann, 2017). We found BRI routes within 25 km of important protected areas including the Dong Phayayen-Khao Yai Forest Complex in Thailand, a UNESCO World Heritage site, the Khao Sok National Park-Khlong Saeng Wildlife Sanctuary landscape, a relic of the oldest remaining rain forest in the world, and the Nam Et-Phou Louey, which was the last site for a breeding population of tigers in Indochina (Laos, Cambodia, and Vietnam) (McCann, 2017). Laurance et al. (2015a) have urged, such wilderness regions and remnants of intact forests with rare and endangered biodiversity should be kept road-free and the development of infrastructure should be strictly prohibited.

4.2. Does BRI traverse important terrestrial ecosystems?

The analysis of potential impacts of proposed BRI infrastructure

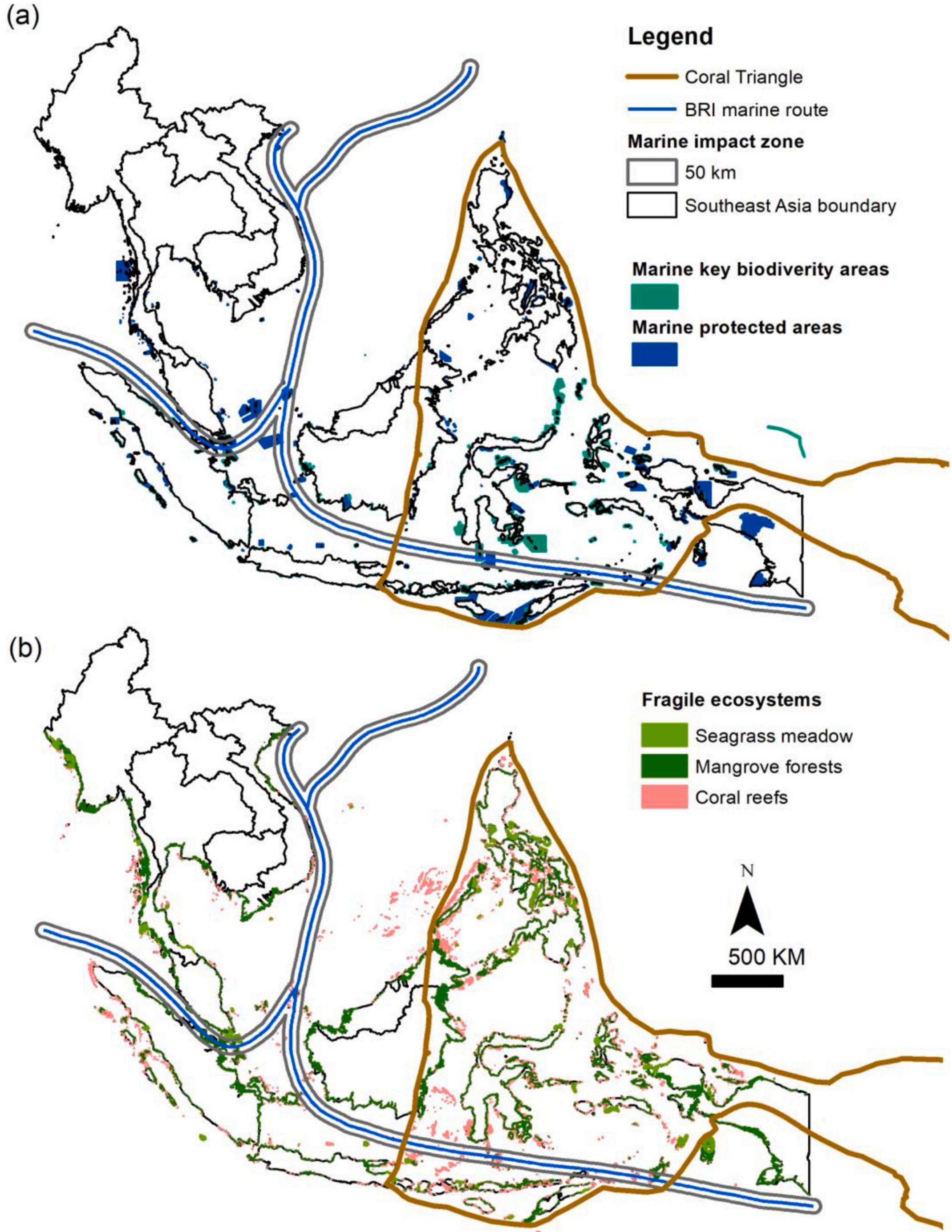

Fig. 9. Map of BRI marine routes representing the Maritime Silk Road overlaid on the marine environment of Southeast Asia, namely (a) protected area and key biodiversity areas; (b) ecologically sensitive ecosystems such as seagrass meadows, mangroves and coral reefs. BRI marine routes represent shipping routes and the 50 km impact zone represents potential negative impacts along these routes.

routes and route types on biodiversity found that BRI is expected to directly bisect 2.3% (11 out of 472) of totals PAs in mainland Southeast Asia of high conservation relevance (IUCN Class I-IV), putting these protected habitats at risk of fragmentation and degradation. Among mainland Southeast Asian countries, PAs in Vietnam, Thailand, and Myanmar, in that order, experience the greatest potential threats from BRI. BRI also overlaps with KBAs and endangered ecosystems. A total of 21 PAs and 14 KBAs are within 5 km of new rails, which are also commonly found within relatively undisturbed areas (Figs. 1a, 2c, 3c). Within 25 km of BRI infrastructure, 71 KBAs have little, none, or unknown legal protected status. In the case of rail impacts on PAs and KBAs, the concern is much higher at closer distances (e.g. buffer zones 1 to 25 km), since rails do not promote accessibility as much as roads do. None of the ecoregions in mainland Southeast Asia have reached a Half-Protected status and 12 Nature Imperilled ecoregions occur in the vicinity of BRI development, representing the few remaining natural habitats in these ecoregions.

PA status may not guarantee protection as PA downgrading, downsizing, and degazettement (PADDD) may occur as a means for governments to support infrastructure development and the range of natural resource exploitation that follows such as mining, logging and wildlife poaching (Hance, 2018). In Southeast Asia, PAs in Vietnam,

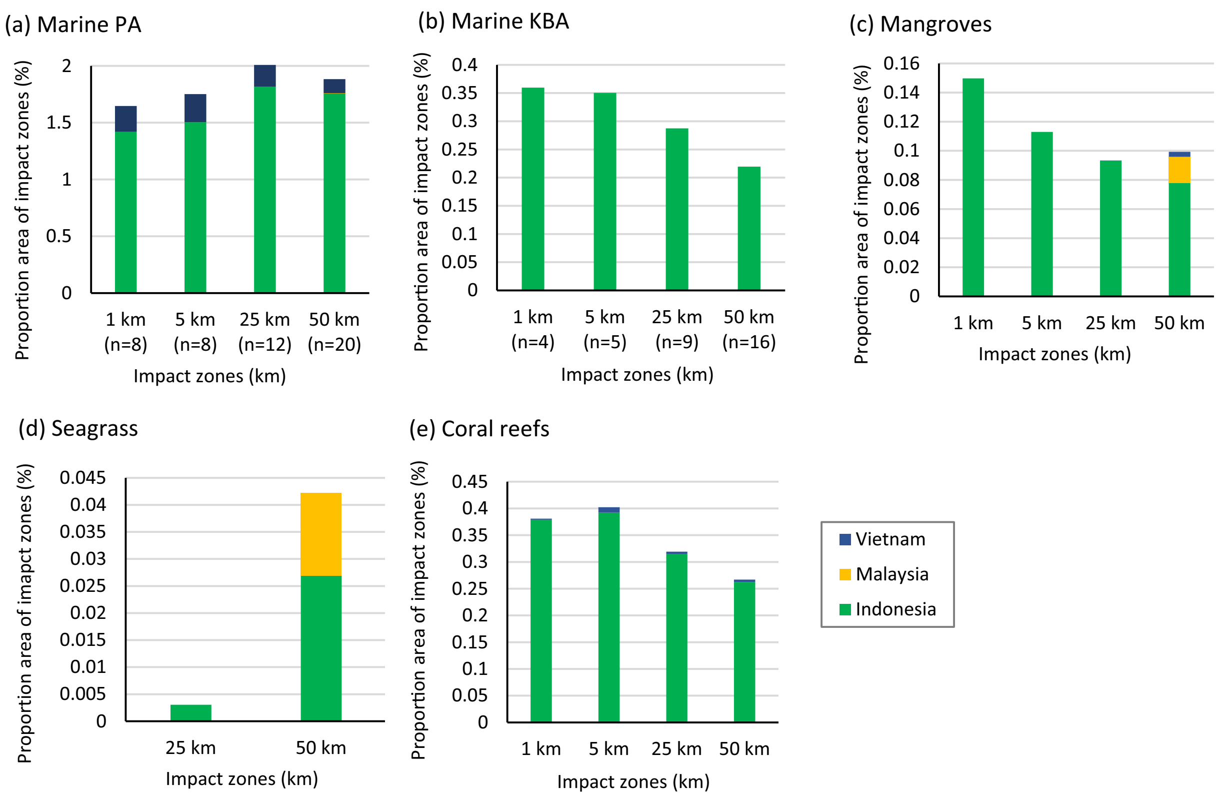

Fig. 10. Proportion area of impact zones that overlap with marine ecosystem indicators by country: (a) Marine PA (b) Marine KBA, (c) Mangroves, (d) Seagrass, and (e) Coral reefs. (a) – (b) Number of marine PAs and KBAs affected in brackets on x-axis.

Cambodia, and Malaysia have a higher probability of undergoing PADDD events compared to Indonesia, the Philippines, and Thailand, which had lower probability than the global mean (Symes et al., 2016). In Peninsular Malaysia, between 2000 and 2010 it was estimated that forests which have undergone PADDD had a 240% higher deforestation rate compared to protected forests and 7% higher deforestation rate than those of non-protected forests (Forrest et al., 2015).

4.3. Does BRI traverse threatened terrestrial species ranges?

Our analyses show the presence of threatened species within the proximity of BRI infrastructure. Within the 5 km buffer, BRI infrastructure overlaps with the ranges of 196 threatened species (81 mammals, 65 birds, 15 amphibians, and 35 reptiles). New rail's 5 km impact zone overlapped with the ranges of 142 threatened species (61 mammals, 54 birds, 6 amphibians, and 21 reptiles), including 26 critically endangered species. Linear infrastructure can fragment their habitat resulting in habitat patches too small to sustain viable populations in the long term, leading to local extirpations and increasing the risk of global extinction. Threatened amphibians are the most vulnerable vertebrate class in our analyses (i.e. compared to birds and mammals). This is because species with restricted distributions or small population sizes experience greater barrier effects and are more susceptible to extinction due to their sensitivity to stochastic changes (Jochimsen et al., 2004). Without careful planning of infrastructure corridors, the population of these species can become highly fragmented and genetically isolated, thus threatening their long-term survival.

The conservation of tropical biodiversity relies on protected areas as an essential element for safeguarding forest-dependent species with large home-ranges such as the Asian elephant (*Elephas maximus*) and tigers (*Panthera tigris,* Gardner et al., 2009). In Sumatran Indonesia, the recently discovered yet critically endangered Tapanuli orangutans (*Pongo tapanuliensis*) (Nater et al., 2017) are threatened by hydroelectric power plant and dam that sits directly in their critical habitat, fragmenting the only habitat with a viable population of 500 individuals (Sloan et al., 2018). Dams and other supporting infrastructure are another overlooked aspect of BRI as new developments, and new areas opened to agriculture are liable to require both power and irrigation and this is likely to have a wide suite of impacts on terrestrial and aquatic systems (Ascensão et al., 2018; Hughes, 2017a, 2017b; Lechner et al., 2018).

4.4. Comparison of impacts by route type and implications for conservation

Certain specific segments of all BRI route types were found traversing forested areas and areas with low human footprint as well as high biodiversity value such as the Karst ecosystems in Vietnam. These areas are typically more vulnerable to drastic, road-induced land use changes and deforestation than are croplands, grasslands, or other non-forest areas (Ledec and Posas, 2003). This is particularly true of tropical rainforests where economic development and ecosystem conservation often clash (Pfaff et al., 2007). For interpreting our analysis, buffer distances close to BRI routes, new infrastructure and roads rather than rail are likely to have greater impacts.

The interpretation of the impact zones is complex and actual impacts will vary with linear infrastructure type, mitigation measures, and the unique characteristics of a particular development. For example, the Kunming-Vientiane trainline is unique in Southeast Asia due to being located in particularly mountainous terrain. It will reportedly include over 62% of its length as bridges and tunnels (Yap, 2017), which means ecological impacts such as the barrier effect, fragmentating landscapes and other impacts from noise and slopes destabilisation from this project may be relatively lower than other projects. Every project will also be subject to differing levels of scrutiny, regulation and enforcement which may differ greatly between jurisdictions and countries (Lechner et al., 2019). Even though in China there are some good examples of high speed rail development with best practice mitigation measures, how BRI projects impact on the environment are likely to be as much a property of the host country regulations, as efforts by Chinese financiers and the Chinese government.

4.5. How will the Maritime Silk Road impact marine ecosystems?

Southeast Asia is a major hub for shipping traffic, with several megaports and extensive networks of shipping lanes. Southeast Asia is China's largest trade partner and accounts for half of China's trade with BRI countries (Songwanich, 2018). Increased marine traffic from BRI is likely to increase risks to endangered and sensitive ecosystems such as mangrove forests, seagrass beds, and coral reefs in the South China Sea. Impacts may include overfishing, coral damage through direct damage from shipping, pollution and the exacerbation of the movement of invasive species (Molnar et al., 2008; Todd et al., 2010). These coastal ecosystems suffer similar threats from anthropogenic activities such as land reclamation, dredging, pollution and sedimentation from urban runoffs, and overexploitation (Yaakub et al., 2014); impacts to individual ecosystems could cause cascading effects from one to another. Our results estimate that as much as 1026 km^2 of mangrove habitats, 439 km^2 of seagrass habitats and 2780 km^2 of coral reefs as well as a significant number of marine PAs and KBAs, 20 and 16, respectively, overlap with BRI marine routes. Unlike the terrestrial routes which have physical dimensions, projecting the footprint of impacts is much more complex due to the changes in future sea traffic associated with new ports and changes in global supply routes associated with the BRI. More complex modelling approaches are needed to be applied to provide greater precision around future BRI sea-traffic (See Lee et al. (2018) for a review of methods for modelling sea transport and maritime logistics).

Construction and damage to coastal mangroves to create new ports has a slew of environmental and socio-economic consequences, not only do these regions provide coastal buffers in the case of tsunamis and other tidal events (Guannel et al., 2016), they also provide nurseries for marine fish (Hughes, 2017a, 2017b). A recent global analysis of BRI port development found that over 400 threatened marine species could be affected (Turschwell et al., 2020). Nearshore habitats are critical for endangered species such as dugongs (*Dugong dugon*) and green sea turtles (*Chelonia mydas*) which are almost completely seagrass dependent (Fortes, 1988). These ecosystems, especially those at the interface between marine and terrestrial areas often fall through the gaps in protection efforts, yet are critical systems for many species and key to the survival of many species such as on the East-Australian Asian flyway (Li et al., 2019).

Marine-based activities that threaten coral reefs include pollution from ports, oil spills, ballast and bilge discharge, garbage and solid waste dumping from ships, and direct physical impacts from groundings and anchor damage (Burke et al., 2002). In 1975, a loss of hundred hectares of mangroves in the Sumatran East Coast was recorded as a consequence of an oil spill (Fortes, 1988; Kirkman and Kirkman, 2002). Areas where new ports are proposed or expanded could result in direct loss of mangrove or seagrass habitats during construction, such is port-related activities such as dredging, shipping movements or recreational boating (Yaakub et al., 2014). Proposed BRI ports include Malacca's deep-sea port in Malaysia, a deep-water port in the Bay of Bengal in Myanmar, a port in Sihanoukville, Cambodia and the expansion of Laem Chabang deep-water port in Thailand as part of the Eastern Economic Corridor (Songwanich, 2018).

4.6. Solutions and mitigations for BRI

BRI could exacerbate existing environmental issues facing this diverse region such as deforestation, climate change, and biological invasion (Sodhi et al., 2010; Hughes, 2017a, 2017b; Morand et al., 2017). In Southeast Asia, BRI is increasingly seen as a threat to the already imperilled biodiversity, given the scale of its infrastructure footprint on frontier ecosystems and endangered biodiversity in the region. Furthermore, 45% of BRI funding is for projects in Southeast Asia (Kong et al., 2019), making the Southeast Asia BRI corridor the most heavily funded of all BRI corridors. Thus, there is even a greater urgency in this region to address and plan for BRI impacts. To reduce its negative impacts on the environment, BRI needs to adopt a rigorous plan to safeguard vulnerable ecosystems and biodiversity.

In addition to avoiding areas of high biodiversity value wherever possible, BRI need to direct a proportion of its investment towards green or ecological infrastructure. Developers from BRI should engage with local authorities in early stages of planning to mitigate impacts of linear infrastructure which are often the precursors for forest conversion and poaching (Clements et al., 2014). BRI could also establish a network of protected areas and wildlife corridors across Eurasia, especially in needed regions such as Southeast Asia. BRI could achieve this by pushing financiers to have better biodiversity safeguards (Narain et al., 2020) and work proactively with environmental specialists and local authorities (Laurance et al., 2015a) such as through the creation of a "green fund". Such a funding mechanism could be used to offset impacts and promote cooperation on expanding existing national initiatives such as Malaysia's Central Forest Spine (Regional Planning Division Department Of Town and Country Planning Peninsular Malaysia, 2009) or Bhutan for Life (National Biodiversity Strategies and Action Plan of Bhutan, 2014). Moreover, BRI could promote transboundary and cross-border conservation activities such as establishing conservation parks and multinational joint research programs, as seen in the Heart of Borneo (Sloan et al., 2019a, 2019b) and to assess the impacts of BRI across biodiversity hotspots (Yang et al., 2016). Such initiatives are indeed highlighted as a component of the science plan which forms a core component of the Digital Belt and road, to ensure the science needed to mitigate impacts of the belt and road is available and accessible (http://www.dbeltroad.org/) (Hughes et al., 2020). Strategic and project-based environmental assessments should be applied at the feasibility stage prior to investment, as well as adopt the mitigation hierarchy to ensure a net gain of biodiversity (Ascensão et al., 2018; Hughes, 2019; Lechner et al., 2018).

BRI can drive the improvement of institutional and legal frameworks for natural source conservation, identify new high-priority habitats while improving and expanding protected areas, secure state-of-the-art advice for specific conservation issues, as well as leverage funds for conserving otherwise unprotected habitats (Quintero, 2007). Using its geopolitical weight, China could make BRI a catalyst for advancing conservation actions that might not otherwise be a country priority. Such an approach (Hughes et al., 2020; Lechner et al., 2018) requires the multi-stakeholder participation of governments, and intergovernmental organisations such as the United Nations Developing Programme, financial institutes, conservationists and the civil society, to realise this vision, and indeed the vision proposed by "ecocivilisation".

One of the foundations for addressing BRI impacts is good spatial planning; thus identifying routes, projecting future impacts and identifying mitigation measures on the ground is fundamentally important. There is a need for greater clarification around proposed BRI routes and the development of a framework to understand and project impacts (Teo et al., 2019). There are no official publicly accessible BRI route maps, meaning that the results of our assessment are likely to, in part, be driven by the choice of BRI routes characterised by the map we chose. Spatial planning needs to be dynamic as the list of planned projects are constantly evolving. For example, in Malaysia two confirmed BRI projects were originally rejected by an incoming new government, but now will go ahead after some revision (Lechner et al., 2019), other projects are likely to change, and evolve, but many formerly intact landscapes.

As well as uncertainty in the routes, uncertainty associated with the biodiversity data and modelling of indirect impact processes require greater research. There is a general lack of research on the effects of road and rail on wildlife especially for rail impacts and especially within Southeast Asia. A recent systematic review of linear infrastructure impacts on wildlife by Popp and Boyle (2017) identified 276 papers and none of those were in Southeast Asia and only 17 on railway wildlife impacts (note: this assessment is based on Popp and Boyle's (2017) supplementary data). Our model characterised impacts based on impact zones however indirect impacts are likely to be more complex and underestimated. For example, the limestone rat (*Niviventer hinpoon*) was found to have >32% of its range intersected at 25 km (Table S4), but this may be an underestimate as, like many other limestone endemic species, the limestone rat may lose additional habitat for the production of cement used for road and associated infrastructure (Hughes, 2019).

Within China huge levels of funding have gone into projects which aim to minimise negative environmental consequences of BRI, more attention is needed to ensure such provisions are applied to all parts of the belt. Given the lack of available biodiversity inventories in many countries, understanding the potential impacts of parts of the route are hugely challenging. The lack of funding in many host countries and lack of capacity have formerly inhibited the development of biodiversity inventories, thus joint funding and “belt and road fellowships” which target capacity building to enable better understanding of biodiversity and the development of sustainable approaches to development are starting to address the information and capacity deficit which currently makes planning for BRI in developing nations difficult.

5. Conclusion

China has already taken substantial steps both to reduce the impact of development on its native biodiversity, and to develop policies (i.e. ecological redlines; Bai et al. (2018)) to maintain diversity and ecosystem service provision. A consequence of this has been said to be the outsourcing of biodiversity losses (Többen et al., 2018). Without careful planning, BRI could considerably add to this outsourcing. The combined use of green policies on China funded projects and sensitivity in developing areas of high ecological value could considerably reduce BRI's negative impacts on the region's biodiversity.

Here we outline regions of sensitivity, and how they differ across Southeast Asia based on a number of metrics, which clearly show that sensitivity varies spatially depending on what element of biodiversity is considered. Development in many of the most important or sensitive regions should be avoided and for all routes careful planning and sensible policies around environmental impact assessment and environmental planning on a project-by-project basis and regionally is required for Southeast Asia - a global biodiversity hotspot. The development of associated infrastructure, both in the immediate vicinity and in the production of raw materials will also need careful planning to avoid further associated ecological damage to native ecosystems and species.

Ultimately, the impacts of BRI depend on how policies are implemented, both in the development of core, and associated infrastructure, and how that infrastructure is implemented with regards to inhibiting human access to fragile regions and to enable wildlife to safely bypass parts of the route. Without overarching policies to mitigate potential losses, implementation is liable to be haphazard and fall to the whims of individual developers and their stakeholders to decide (and are therefore liable to prioritise profit, rather than environmental or social impacts); and thus overarching approaches will be necessary to ensure BRI can be implemented without long term costs to local communities and biodiversity along its route.

Authors declaration

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary information

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