



CHAPTER 3



Ecological impacts of extractive industries on ape populations

Introduction

This chapter explores the significant threats and risks to apes, and their habitat, that result from the activities of extractive industries. All apes are protected by national and international laws throughout their geographic range. It is therefore illegal to kill, capture, or trade in either live apes or their body parts. It is important to understand where and how extractive industries affect great apes and their habitat during each phase of a project. In mining, oil, and gas projects (Chapter 5), these phases include exploration and evaluation, preliminary engineering and alternatives analysis, final engineering and site selection, construction and commissioning, operation, closure, and post-closure phases. All phases of all

extractive industries are likely to have some impact on resident apes, although the scale and severity are likely to vary. Generally speaking, the behavior and physiology of wildlife are known to be impacted by human activities (Griffiths and van Schaik, 1993; Kinnaird and O'Brien, 1996; Woodford, Butynski, and Karesh, 2002; Blom *et al.*, 2004a; Wikelski and Cooke, 2006; Rabanal *et al.*, 2010; Ruesto *et al.*, 2010; Chan and Blumstein, 2011). Species' responses to environmental disturbance will, however, vary according to their biological dispositions and the type and scale of disturbance. For example, species with highly specialized requirements may manifest significant adverse impacts, as found in studies looking at the impact of logging on terrestrial and bark-gleaning insectivorous birds or bats, while those with more general requirements may be less affected (Putz *et al.*, 2001; Peters, Malcolm, and Zimmerman, 2006).

The list of potential impacts of extractive industries on ape populations is extensive and diverse: (1) Habitat loss from large-scale clear-fell logging and opencast mining will result in total loss or displacement of resident ape populations. (2) Habitat disturbance and degradation from selective logging, subterranean, and smaller-scale mining operations will likely impact the home range and resource use of resident apes, potentially resulting in additional knock-on effects. Changes in resource abundance could, for example, drive changes in activity patterns and energy budgets. These changes may be adaptive, but in some circumstances lowered energy budgets may lead to increased mortality through starvation, stress, and lowered fertility, ultimately reflected in lowered carrying capacity in affected habitats. Indeed, reduced population densities in forests degraded through selective logging are a common theme discussed below. Habitat fragmentation caused by infrastructure development and general reduction in forest

quality may also have long-term effects, including the isolation of sub-populations and a reduction in long-term population viability (see Box 3.1). Social effects may also be expected as habitats are impacted, forcing groups into neighboring areas and increasing contact with conspecifics, potentially causing loss of social cohesion in groups and increased aggression, conflict, and mortality. All of these factors may also increase levels of stress on ape populations with impacts potentially including altered energy budgets, changes in social behavior, higher mortality rates, immunosuppression, lowered growth rates, and reduced reproductive success (Woodford *et al.*, 2002; Wikelski and Cooke, 2006).

In addition to the direct impacts of extractive operations, some impacts will be indirect consequences of other subsistence or commercial activities that have been put in place as a result of the work or economic activity generated by extractive industries. The often more significant indirect impacts result from the opening up of forests to people, driven by increased population size and wealth, and accessibility (to forests and markets) through the development of transport routes into once remote areas. Threats that are indirectly associated with logging and other extractive industries include increased targeted hunting (i.e. poaching) of apes and indirect hunting, where other species are targeted but apes are unintentionally caught and killed. This is for commercial and subsistence bushmeat consumption, perceived medicinal properties, and live animal trade. Further habitat degradation and fragmentation, land conversion for agriculture, the potential introduction of human diseases, and increased spread of diseases between resident apes can adversely affect their populations (Chapter 7). Forests overly degraded by timber extraction or mining become more prone to drought and fires, and other stochastic events, which can in turn have disastrous

consequences for ape survival. Increasingly, these direct and closely linked indirect consequences are further intensified by the cumulative impacts resulting from multiple industries and activities operating within the same landscapes (Chapter 7).

Despite the large list of potential impacts that extractive industry operations may have on apes, many are speculative in that causal links have not been demonstrated. However, we can extrapolate from what is known about

the processes of extraction and the extensive information available on ape socioecology (see “Potential long-term impacts and future studies” on page 93). Other impacts have been documented in the relatively small number of studies that have followed ape populations from pre-extraction to post-extraction. In this chapter, we draw on a large body of literature to come to conclusions about ape responses to the activities of extractive industry. We address the issues

BOX 3.1

Can great apes survive in forest fragments?

As Southeast Asia’s forests are cleared, orangutans are seeking refuge in surrounding areas. They may return as forest regenerates, but degraded forests do not meet all of the orangutan’s biological requirements. They need a mosaic of habitat types, as in the highly fragmented Kinabatangan floodplain of Malaysia, where riparian and mixed lowland dipterocarp forest can still be found along the riverbanks (Ancrenaz *et al.*, 2010). Orangutans are also known to persist in acacia and eucalyptus plantations (Meijaard *et al.*, 2010), although the long-term viability of these individuals is uncertain.

In Africa, great ape populations outside the central basin are greatly threatened by habitat fragmentation, and much of East and West Africa has been deforested by human activities, principally slash-and-burn agriculture (e.g. Brncic, Amarasekaran, and McKenna, 2010). Chimpanzees and bonobos are capable of occupying a wide range of habitat types, so they are not confined to dense forest. Chimpanzees inhabit mosaics of savanna–woodland, gallery forest, and relatively impoverished dry forests in Guinea, Mali, Senegal, and Tanzania; some bonobos occur in mosaics of swamp forest, dry forest, marshy grassland, and savanna–woodland. Nonetheless, chimpanzees and bonobos are heavily dependent on any available tree cover for shade and nesting in these open environments. In Gabon, chimpanzee densities have been found to be similar in fragmented forest patches and swaths of continuous forest, whereas gorilla densities were much lower in fragmented than in continuous forest because of their general reluctance to cross large unforested gaps (Tutin, White, and Mackanga-Missandzou, 1997).

Studies of habitat fragmentation as a result of logging suggest that the impacts on great apes depend on the species (Tutin and Fernandez, 1984; Plumptre and Reynolds, 1994; Hashimoto, 1995). Onderdonk and Chapman (2000) studied primate occupation and the characteristics of forest fragments outside Kibale National Park, Uganda. Evidence of chimpanzee presence was found in 9 of 20 fragments, some as small as 0.008 km² (less than 1 hectare). However, the authors had

the impression that chimpanzees were foraging in these localities for short periods only and frequently moved between patches. They did not find a relationship between primate presence and specific patch characteristics (size of the patch, distance to the next nearest patch, distance to the national park, or number of food trees present). Chimpanzees in Bulindi, also in Uganda, survive in fragmented riverine habitat comprised of markedly different food sources to those in nearby Budongo. Apparently, those resources are sufficient for the chimpanzees to survive and may even be a direct result of persistent human disturbance (McLennan and Plumptre, 2012). Similarly in Gabon, chimpanzees and gorillas visited natural forest fragments but did not continuously occupy these small patches of forest, which were surrounded by savanna grasslands (Williamson, Tutin, and Fernandez, 1988; Tutin, 1999).

A recent survey in Sierra Leone (Brncic *et al.*, 2010) revealed that approximately 2000 chimpanzees are living outside officially protected areas, travelling between the remaining forest patches, feeding in regenerating farmbrush and secondary forest, but relying heavily on crops grown for human consumption. It is not yet clear if these individuals will survive into the long term or if they are remnants of a dwindling population. Chimpanzees seemed to have managed to survive in fragmented forests in Nigeria, but sites surveyed recently are losing their remaining chimpanzees (Greengrass, 2009).

According to Harcourt and Doherty (2005), 65% of forest fragments where primates are found have an area of less than 1 km², which is too small to support great apes in the long term unless connected to other suitable habitats. These habitats can be natural or human modified, such as the forest–farm mosaics that are typical of East and West Africa and frequently used by great apes (Hockings and Humle, 2009). The critically endangered Cross River gorilla persists in a largely fragmented landscape; however, habitat and dispersal corridors are extensive (Bergl *et al.*, 2012). So it seems that great apes in modified habitats are dependent on resources elsewhere in the landscape, and that habitat connectivity via networks of forest corridors must be maintained if they are to survive.

faced by apes from the extractive industries of logging and mining separately. We also split analysis along taxonomic and geographical lines, and consider the great apes – orangutans and African apes (gorillas, chimpanzees, and bonobos) – and gibbons separately owing to their differing ecological requirements and threats posed by different extractive industries and regional standards. We begin by describing the apes' socioecology to provide a backdrop to the documented and potential ecological impacts on these species. We then review studies that have detailed the impacts of extractive industries on apes and speculate on impacts that additional study may reveal.

Key findings:

- Clear felling is incompatible with ape persistence and it results in their total absence.
- Ape tolerance of selective or responsible logging is not fully understood, but over-harvesting of timber can lead to a significant reduction in population densities.
- Changes in ape behavior as a result of logging are poorly understood, but could lead to a negative energy balance in apes in logged forest owing to changes in the availability of food.
- Clear themes on the impacts of logging on gibbon persistence are difficult to isolate, especially given the family's large geographic range.
- Crucial information on the impacts of mining on all apes is lacking.
- There is a clear and pressing need for education in extractive industries, so that they understand the importance of early stage (baseline) ape population studies.
- There is a need for legal requirements in all ape countries to adopt wildlife-friendly best practices before, during, and after exploration/extraction have occurred.

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Ape socioecology

Great ape socioecology

There are six species of great ape: two orangutans (Bornean and Sumatran), two gorillas (eastern and western), the chimpanzee, and the bonobo. Here we present an overview of the aspects of great ape socioecology and the basic requirements for their survival that are important in the context of this book. There is considerable variation among species and even among populations of the same subspecies. For more detailed information on orangutans see the volume by Wich *et al.* (2009b) and for recent syntheses on African great apes see Emery Thompson and Wrangham (2013), Reinartz, Ingmanson, and Vervaecke (2013), Williamson and Butynski (2013a, 2013b), and Williamson, Maisels, and Groves (2013).

Social organization and structure

Social organization differs considerably among the three great ape genera: orangutans are semi-solitary, gorillas live in stable mixed-sex groups, and chimpanzees and bonobos form dynamic (fission–fusion) communities. The chimpanzee and bonobo communities are multi-male/multi-female closed social networks, which fission into smaller parties according to food availability and presence of cycling females (e.g. Wrangham, 1986), or come together (fusion) at large food sources. The average size of a chimpanzee community is 35 individuals, although one especially large community of 150 members is known in Uganda (e.g. Mitani, 2009). In forest habitats, party size is usually 5–10 individuals; in the savanna–woodlands of Fongoli, mean party size is 15 (Pruetz and Bertolani, 2009). Bonobo communities comprise 10–120 individuals. When foraging on the ground, bonobo social units splinter into mixed-sex parties that are larger and more cohesive than chimpanzee parties,

averaging 5–23 individuals. In both species, party sizes tend to be smaller when fruit is scarce (e.g. Mulavwa *et al.*, 2008).

Large body size and folivorous (leaf-eating) tendencies enable gorillas to cope with fruit shortages and reside in cohesive social units. Gorillas live in relatively stable groups with one or more adult “silverback” males, several females, and their offspring. Group size, composition, and patterns of dispersal are similar across all gorilla taxa; median group size of both species is 10 individuals. One of the main roles of the dominant male is to use his strength, size, and intimidating displays to defend females from other males. Among female great apes, only gorillas live in permanent association with males, relying on males to protect their infants against infanticidal attacks by other males (Robbins *et al.*, 2004). A female who transfers to another group with an infant faces the risk of her offspring being killed by the dominant male in her new group (Watts, 1989; see also “Reproduction”).

Orangutans have loosely defined communities in which residents are familiar with other orangutans in their neighborhood. Most flanged adult male orangutans lead a semi-solitary existence, while the smaller unflanged adult males are comparatively tolerant of other males (some adult male orangutans increase in size and develop cheek flanges, linked to increases in testosterone levels (Emery Thompson, Zhou, and Knott, 2012)). Adult female orangutans are more gregarious than adult males and related females sometimes travel together. Unflanged males will travel with females and offspring, and this gregariousness significantly impacts their habitat requirements and ranging behavior. The forests of Sumatra are more productive than on Borneo (Wich *et al.*, 2011c) and Sumatran orangutans congregate when food is abundant (Wich *et al.*, 2006). Sumatran orangutans also have slightly larger party sizes (1.5–2.0 individuals; Mitra Setia *et al.*, 2009).

Reproduction

Great apes reproduce very slowly. Gestation length in gorillas and orangutans is about the same as for humans, i.e. 9 months; it is slightly shorter in the smaller chimpanzees and bonobos at 7.5–8.0 months. Females usually give birth to just one infant at a time, although twin births do occur. In those cases, it is often not possible for the mother to keep both infants alive (e.g. Goossens *et al.*, 2011). There are no birth seasons; however, because the female’s reproductive cycle is energetically demanding and requires her to be in good health, conception will be determined by food availability and this may be seasonal (Emery Thompson and Wrangham, 2008). Number of births may peak during particular months in relation to resource availability. Bornean orangutans living in highly seasonal dipterocarp forests are most likely to conceive during mast fruiting events, when seeds high in fat are plentiful (Knott, 2005). Sumatran orangutans do not face such severe constraints (Marshall *et al.*, 2009a). Gorillas are somewhat less dependent upon fruit and there is no seasonality in their reproduction. However, chimpanzee and bonobo females are more likely to ovulate when fruit is abundant, so in some populations there are peaks in numbers of females conceiving, with contingent peaks in birth rates (e.g. Anderson, Nordheim, and Boesch, 2006).

Young great apes develop relatively slowly and are dependent on their mother for several years, sleeping in her nest either until they are weaned or the next sibling is born. Much of what is known about the age at which weaning is completed is preliminary, but estimates range from 4–5 years for African apes, 5–6 years for Bornean orangutans, to 7 years for Sumatran orangutans. Weaning marks the end of infancy for African apes, but orangutan infants do not become fully independent of their mothers until 7–9 years of age (van Noordwijk *et al.*,

2009). Resumption of a female's reproductive cycle is inhibited by lactation, so while her infant is nursing, she cannot become pregnant (e.g. Stewart, 1988). As a result, births are widely spaced, averaging 4–7 years in African apes, 6–8 years in Bornean orangutans, and 9 years in Sumatran orangutans. The orangutans' exceptionally long interbirth intervals are thought to be a consequence of their more solitary lifestyle. This investment by orangutan mothers results in lower mortality and about 90% survival of infants, compared to 73% in mountain gorillas and as low as 50% for some chimpanzee populations, such as those in western Tanzania (Wich *et al.*, 2004, 2009a).

Interbirth intervals can be shortened by a phenomenon common throughout the animal kingdom and significant in the context of behavior resulting from external impacts: infanticide is the killing of unweaned offspring by a member of the same species (Harcourt and Greenberg, 2001). In great apes this is typically an unrelated adult male and results in early resumption of the mother's reproductive cycle (since the infant is no longer suckling). Infanticide has been documented among gorillas and chimpanzees, but has not been observed in orangutans – due in part to their more solitary lifestyle (Beaudrot, Kahlenberg, and Marshall, 2009). Some female great apes adopt tactics to “create confusion” about paternity by mating with multiple males. Bonobo males have no indication of whether or not they sired any particular offspring, and infanticide seems to be absent in their communities (Furuichi, 2011).

Slow rates of reproduction are common to all great apes, due to the mother's high investment in a single offspring and the infant's slow development and maturation. Male bonobos reach sexual maturity by 10 years of age, and male chimpanzees mature between the ages of 8 and 15 years (Emery Thompson and Wrangham, 2013). Male eastern gorillas mature at 15 years;

male western gorillas reach full maturity at 18 years (Breuer *et al.*, 2009). Male orangutans reach sexual maturity between the ages of 8 and 16 years, but may not become flanged until they are at least 35 years old (Wich *et al.*, 2004). Orangutans and gorillas are among the most sexually dimorphic of primates, reflecting intense physical competition between adult males. Some flanged male orangutans are extremely aggressive and are able to monopolize an area into which they attract receptive females (Delgado, 2010).

Female great apes reach maturity at similar ages: orangutan females begin to display sexual behavior at 10–11 years, chimpanzees 7–8 years, bonobos start cycling at 9–12 years, gorillas at 6–7 years. Age of giving birth for the first time in orangutans is 15–16 years, 10 years in gorillas (range of averages 8–14 years), 13.5 years in chimpanzees (mean at different sites 9.5–15.4 years), and 13–15 years in bonobos. Mean birth rate in gorillas and chimpanzees is 0.2–0.3 births/adult female/year, or one birth per adult female every 3.3–5.0 years. Female lifetime reproductive success has been estimated for mountain gorillas and chimpanzees: on average, chimpanzee females give birth to four offspring during their lifetime, but only 1.5–3.2 survive beyond infancy (e.g. Sugiyama and Fujita, 2011); mountain gorilla females produce an average of 3.6 offspring during their lifetime (Robbins *et al.*, 2011). Orangutans have the slowest life history of any mammal, with later age at first reproduction, longer interbirth intervals, and longer generation times than the African apes (Wich *et al.*, 2009a). Generation time in the great apes is between 20 and 25 years (IUCN, 2013).

Habitat preferences and nest building

Most great apes live in closed, moist, mixed tropical forest, and they occupy a range of

forest types, including lowland, swamp, seasonally inundated, gallery, coastal, submontane, montane, and secondary regrowth. Eastern and western chimpanzees also occur in savanna-dominated landscapes. The largest great ape populations are found below 500 m elevation in the vast *terra firma* and swamp forests of Africa and Asia (e.g. Morrogh-Bernard *et al.*, 2003; Stokes *et al.*, 2010) although eastern gorillas range up to 3800 m altitude. Gorillas, chimpanzees, and bonobos are rarely found in monodominant stands of *Gilbertiodendron dewevrei* where the herb layer is sparse, except during mast fruiting that occurs every 4–5 years in Central Africa (e.g. Blake and Fay, 1997).

African great apes are semi-terrestrial. Orangutans have been assumed to be almost exclusively arboreal, but recent studies show that Bornean orangutans use terrestrial locomotion (Loken, Spehar, and Rayadin, 2013). Nevertheless, orangutans are not

adapted to travel on the ground and they depend more heavily on lianas to help them move through the canopy without descending to the forest floor than the other great apes (Thorpe and Crompton, 2009). Great apes not only feed but also rest, socialize, and sleep in trees, although gorillas and chimpanzees often rest on the ground during the daytime. Being large-brained mammals, they need to sleep for long periods. A behavior that is partially innate to all great apes is that they build nests to spend the night in; each weaned individual makes a new nest almost every night (e.g. Tutin *et al.*, 1995). Gorillas often nest on the ground, building cushions of vegetation, usually from herbs. In some populations, chimpanzees occasionally sleep on the ground (e.g. Koops *et al.*, 2007). To build nests, great apes need access to trees sturdy enough to support their weight, yet flexible enough that the branches can be bent and secured, and with abundant foliage to provide a cushion against hard surfaces.

Photo: To build nests, great apes need access to trees sturdy enough to support their weight, yet flexible enough that the branches can be bent and secured, and with abundant foliage to provide a cushion against hard surfaces.
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“The great apes’ frugivorous nature is an important factor in maintaining forest diversity as they are important seed dispersers.”

surfaces. These beds are constructed high in the trees, generally 10–30 m above the ground (e.g. Morgan *et al.*, 2006). Orangutans choose to nest in trees with a large diameter and other features that increase stability, such as buttresses, in a position that will offer protection from wind and rain (e.g. Prasetyo *et al.*, 2009; Cheyne *et al.*, 2013).

Nests provide comfort and support that improves the quality of sleep. A recent study comparing the nesting habits of chimpanzees in Senegal and Tanzania has shown that nests have multiple functions, which include providing insulation and simply preventing a fall from a tree while asleep, but that predation is also an important factor for nest-building above ground (Stewart and Pruetz, 2013). Nesting in trees is a way to avoid predators and large forest mammals that are active at night, such as pigs and elephants. Sleeping location is critical for populations vulnerable to poaching: western lowland gorillas in Cameroon and Grauer’s gorillas in eastern DRC (Democratic Republic of Congo) are known to nest at steep locations that humans would find difficult to reach (E.A. Williamson, personal observation). Various anti-parasite and anti-disease functions have been ascribed to nest building, particularly as nest reuse is uncommon (e.g. Fruth and Hohmann, 1996; McGrew, 2010). It is evident, therefore, that the structure of the habitat and diversity of tree species are critical to great apes.

Foods and feeding

Great apes are not strictly vegetarian, as all taxa consume insects and some eat meat; however, they are all adapted to a diet of plant parts that are easy to digest: succulent pulp, new leaves, petioles, buds, shoots, and herbs. Ripe, sugary fruits produced by forest trees are their primary source of nutrition, with the sole exception of mountain gorillas, which live at high altitude where few suc-

culent fruits are available (Watts, 1984). The other African apes average 62–85% fruit in their diet, with marked seasonal variation (e.g. Rogers *et al.*, 2004). Bornean orangutans are less frugivorous than Sumatran orangutans as they experience months when almost no fruit is available (Russon *et al.*, 2009). The great apes’ frugivorous nature is an important factor in maintaining forest diversity as they are important seed dispersers (e.g. Tutin *et al.*, 1991; Gross-Camp, Masozera, and Kaplin, 2009; Beaune *et al.*, 2013).

Even the largest of the apes occasionally climb to heights of 30 m or more when feeding. They do not forage randomly, but are selective feeders, tending to choose items from relatively few of the wide range of foods available (e.g. Leighton, 1993). Although much of their food is harvested in the canopy, African apes forage at all levels of the forest, and most also specialize on the abundant terrestrial herbs that are available all year round in more humid forested areas.

During periods of food scarcity, dietary flexibility is crucial. “Fallback foods” are food items that are always available but which are “not preferred” and are usually poor quality, such as bark and unripe fruit (Marshall and Wrangham, 2007). When succulent fruit is rare, bonobos, chimpanzees, and gorillas eat more herbaceous and woody vegetation, such as shoots, young leaves, and bark (e.g. Rogers *et al.*, 1994); at many sites, chimpanzees eat more figs when preferred alternatives are rare. Similarly, orangutans may consume large quantities of bark and figs, which are produced in abundant crops year-round. Some Bornean orangutan populations live in such highly seasonal habitats that they experience periods of negative energy balance during food shortages (Knott, 1998a, 2005).

Ranging

Great apes travel through the forest in daily searches for food. Their movements are not

random and are generally restricted to a particular location, an area of forest that the ape or group of apes knows well. Foraging in complex forest environments requires spatial memory and mental mapping, and it has been demonstrated that chimpanzees are capable of memorizing the individual locations of thousands of trees over many years (Normand and Boesch, 2009). The other great ape species are likely to possess similar mental capacities.

More or less restricted to the canopy, orangutans do not travel long distances: Bornean adult females and flanged adult males move 200 m to 1 km each day. The lighter and more agile unflanged adult males are able to move faster and usually double the distance. Sumatran orangutans move farther, but still average less than 1 km each day (Singleton *et al.*, 2009). The semi-terrestrial African apes range considerably longer distances and the most frugivorous roam several kilometers each day: chimpanzees 2–3 km, with occasional 10 km excursions; and bonobos and western lowland gorillas average 2 km, but sometimes 5–6 km (e.g. Doran-Sheehy *et al.*, 2004). Habitat and season affect day length as well as home-range use.

The size of the area used habitually by an individual, group, or community (depending on the species) is called the home range. This averages 4–8 km² for male Bornean orangutans, which is small compared with Sumatran males, whose home ranges in swamp forest may exceed 25 km² (Singleton and van Schaik, 2001). Orangutan home-range overlap is usually extensive. High-status flanged males are to some degree able to monopolize both food and females, and so may temporarily reside in a relatively small area (e.g. Delgado and van Schaik, 2000). Establishment of a circumscribed home range helps secure access to resources within it (e.g. Delgado, 2010), and a male's home range may encompass several (smaller) female home ranges. Flanged male orangu-

tans do not tolerate one another, but rather than using active defense, they establish personal space by emitting long calls. Unflanged Sumatran males occasionally congregate around a favored food source where a flanged male may also be present and as long as distance is maintained, physical conflicts are rare; however, close encounters between adult male orangutans trigger aggressive displays that sometimes lead to fights (Knott, 1998b). When males do battle and inflict serious injuries on their opponent, infection of the wounds can result in casualties. Such deaths have been known amongst male Bornean orangutans (Knott, 1998b).

Eastern gorillas range over areas of 6–34 km² (Williamson and Butynski, 2013a). Western gorilla home ranges average 10–20 km², although Head *et al.* (2013) reported a home range size of over 50 km² in coastal Gabon. Gorillas are not territorial and range overlap between neighboring groups is substantial. Encounters between groups using the same area can occur without them being able to see each other, due to the poor visibility in dense forest. Instead, dominant males may exchange vocalizations and chestbeats, sometimes for hours, until one or both groups move away. Groups ignore each other under particular conditions, such as in the large swampy clearings found in northern Congo, where good visibility allows adult males to monitor potential competitors from a safe distance (Parnell, 2002). These males may display, but physical contact between them is rare. In contrast, in a study of mountain gorillas, adult males engaged in contact aggression during 17% of group encounters (Sicotte, 1993). Serious aggression between gorillas is rare, but when contests escalate, fighting can be intense and the outcome fatal. Deaths from septicemia have followed injuries sustained during intergroup interactions (Williamson, in press).

The home ranges of chimpanzees living in forest habitats vary between 7 and 41 km²

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(e.g. Emery Thompson and Wrangham, 2013), but are larger in drier habitats (e.g. over 65 km², Pruettz and Bertolani, 2009). Females have small “core” areas within a community home range defended by the males. Males are highly territorial and patrol the boundaries of their range, especially if it borders that of another community’s range. Groups of males may attack members of neighboring communities and some populations are renowned for their aggression (Williams *et al.*, 2008). Wilson *et al.* (2012) reported that most attacks are launched by communities and patrols with large numbers of males, and that victims are usually adult males and infants. The protagonists benefit by gaining females or increasing the size of their range. Bonobo communities share home ranges of 22–58 km² and the overlap between community ranges is 40–66% (e.g. Hashimoto *et al.*, 1998). Bonobos exhibit neither territorial defense nor cooperative patrolling. Encounters between bonobo parties from different communities are frequent and characterized by high-pitched excitement rather than conflict (e.g. Hohmann *et al.*, 1999). Some encounters are aggressive, but thus far no lethal incidents have been recorded (e.g. Hohmann *et al.*, 1999).

Where gorillas and chimpanzees are sympatric, the two species occasionally meet at the same fruiting trees. In most circumstances, there is dietary partitioning between chimpanzees and gorillas to avoid direct competition over food sources. If the area of available habitat is restricted, such mechanisms for reducing competition will be compromised. Observations of interactions between the two species are rare, and encounters can either be peaceful or result in agonistic contests. In Uganda, a gorilla was seen feeding in a fig tree within a few meters of several adult male chimpanzees, although at the same site a party of chimpanzees temporarily prevented a gorilla group from entering the tree they occupied (Stanford,

2006). Co-feeding has also been witnessed in the Republic of Congo. Aggressive encounters between gorillas and chimpanzees have not been observed and it is thought that both species may be more tolerant when they are mutually attracted to a highly preferred food source, especially in times of fruit scarcity (Morgan and Sanz, 2006).

Two key points to be noted here are:

1. that documenting the biology of these long-lived species takes decades of study due to their slow rates of reproduction; and
2. that a great ape population that has been reduced in size is likely to take several generations to recover.

These factors make great apes far more vulnerable to threats than smaller, faster breeding species. The orangutan’s rate of reproduction is the slowest of all and they are, therefore, the most susceptible to population losses. Also significant is that great apes have large brains and rely heavily on social learning. Populations and individuals exhibit differences in learned behavior and different ways to exploit their natural habitat. Based on these observations, we can expect great apes to adapt to habitat changes to a certain degree and therefore to show some resilience to habitat degradation and exploitation.

Gibbon socioecology

Gibbons (Family Hylobatidae) are the most widely distributed of ape taxa, occurring from Assam, India, eastwards through Bangladesh, Myanmar, Thailand, South-western China, Cambodia, Laos, and Vietnam, and southwards through Malaysia and Indonesia. Currently 19 species in 4 genera are recognized; *Hylobates* which contains 9 species, *Nomascus* the next most speciose with 7 species, *Hoolock* with 2 species, and the monospecific *Symphalangus*

(IUCN, 2013). Indonesia holds the most gibbon taxa with 8, followed by Laos, Vietnam, and China with 6 each. Sympatry between species occurs between some taxa in generally narrow bands with the exception of the ecologically distinct siamang and white-handed gibbon species *Hylobates lar* and *H. agilis*, which may be sympatric.

Gibbons are highly threatened, and have been referred to as the most threatened of primate families (Melfi, 2012) with four species critically endangered, 13 endangered, one vulnerable, and one not yet assessed (*Nomascus annamensis*) on the IUCN Red List of Threatened Species (IUCN, 2013). The urgent nature of this conservation situation has been driven by large-scale habitat loss and fragmentation, and hunting. Drivers for these threats and their relative severity are variable given the wide distribution of the Hylobatidae across ten countries with variable ethnological and legislative environments, levels of forest dependency of rural communities, and commercial forest exploitation. Hunting of gibbons occurs largely for subsistence, traditional Chinese-based medicine, and for the pet trade, while habitat loss and degradation is driven by conversion of forest for small-scale and industrial-scale agriculture, infrastructure development and of specific relevance to this publication, logging and mining operations (see Chapter 7 for more information on indirect impacts).

The Hylobatidae occur across a wide range of habitats, including predominantly lowland, sub-montane, and montane broad-leaf evergreen and semi-evergreen forests, as well as dipterocarp dominated and mixed deciduous forests. Some members of the *Nomascus* genus also occur in limestone karst forests and some populations of the *Hylobates* genus occur in swamp forest. Gibbons may occur from sea level up to around 1500–2000 m asl (above sea level) although this is taxon and location specific. *Nomascus concolor* has been recorded up

to 2900 m asl in China, for example. Being strictly arboreal (Bartlett, 2007) (with the exception of the rarely recorded behavior of moving bipedally and terrestrially across forest gaps or to access isolated fruiting trees in more degraded and fragmented habitats) the Hylobatidae are intimately impacted by the extent and quality of forest.

Gibbons are also reliant on forest ecosystems for sourcing food. Gibbon diets are generally characterized by high levels of fruit intake, with figs dominating in some studies, supplemented with young leaves and, to a lesser extent, mature leaves as well as flowers (Bartlett, 2007; Elder, 2009). Reliance on other protein sources such as insects, birds' eggs, and small vertebrates has been recorded but is likely underrepresented in the literature. The gibbons' frugivorous nature is also significant in maintaining forest diversity as they are important seed dispersers (McConkey, 2000, 2005; McConkey and Chivers, 2007).

Gibbons are territorial, with each family group maintaining a territory defended from other groups. Territories average about 0.42 km² (42 ha) across the family (Bartlett, 2007) but there is considerable variation and there is some indication that the more northerly *Nomascus* taxa may maintain larger territories, possibly related to lower resource abundance at some times of year in these more seasonal forests. Gibbons are also generally typified as forming socially monogamous family groups. More recent studies, however, have revealed they are not necessarily sexually monogamous (Palombit, 1994). Some notable exceptions include extra-pair copulations (mating outside of the pair-bond), individuals leaving the home territory to take up residence with neighboring individuals, and male care of infants (Palombit, 1994; Reichard, 1995; Lappan, 2008). It also appears that the more northerly *N. nasutus*, *N. concolor*, and *N. haianus* commonly form polygynous groups composed of more than one breeding female (Zhou *et al.*, 2008; Fan

“Gibbons are highly threatened, and have been referred to as the most threatened of primate families.”

Peng-Fei *et al.*, 2010; Fan Peng-Fei and Jiang Xue-Long, 2010). There is still no conclusive argument regarding these variable social and mating structures, but they may be natural or a by-product of small population sizes, compression scenarios, or sub-optimal habitats.

Both males and females disperse from their natal groups (Leighton, 1987), at approximately 9 years of age, based on limited data (Brockelman *et al.*, 1998), and set up their own territories. They generally have their first offspring at around the same age. Data from captive settings, however, suggest gibbons may become sexually mature much earlier than this, as early as 5.5 years of age (Geissmann, 1991). Interbirth interval is in the range of 2–4 years, with a gestation period of approximately 7 months (Bartlett, 2007). Although captive individuals have been recorded living upwards of 40 years of age, gibbon longevity in wild conditions is unknown and thought to be considerably shorter. Due to the gibbons' relatively late age of maturation and long interbirth intervals, reproductive lifetime may be only 10–20 years (Palombit, 1992). Population replacement in gibbons is therefore relatively slow.

Studies of the direct impacts of logging on ape populations

Commercial and artisanal logging cause changes in both forest composition and structure, ranging from degradation to elimination of habitat. As forest dependent species, the magnitude of negative impacts on apes is greatest in the case of clear felling as this results in the removal of most if not all trees. Clear felling and ape persistence are incompatible. Since it results in the total absence of apes, we do not consider clear felling in this section and focus instead on selective logging. There are differences between selective logging and responsible

logging (reduced-impact logging (RIL), as described in Chapter 4). Selective logging is a forestry technique devised to mimic at some level natural rates of tree fall through the removal of only a percentage of commercially saleable trees (Okimori and Matius, 2000). Theoretically this allows for the sustainable use of forests, as natural regeneration is allowed before logging is recommenced (Rijksen, 1978). Even at low levels of removal, however, significant damage to forest can be expected, with extraction machinery and falling trees causing additional damage to standing trees (Mittermeier and Cheyney, 1987). It has been variously reported that even with the removal of only 10% of trees in an area, 55% of other trees were lost (Rijksen, 1978), or that with removal of only 3.3% of trees, 50.9% of trees with a diameter at breast height (DBH) ≥ 30 cm were also destroyed (Johns, 1986b).

Over the last 10–20 years, much research has been directed towards understanding the effects of logging activities on wildlife in tropical forests. This is a particularly challenging area of research and it has proven difficult to draw conclusions regarding the impact of specific logging practices that are broadly applicable across the sector. This is partly due to the sheer complexity of life found in tropical forests, compounded by the innate variability between study sites, logging techniques used, species responses, as well as study methods. Disentangling the interactions between these and the potential direct and indirect impacts is problematic. Survey results do not reflect solely the impact of forestry practices, but a myriad of indirect or collateral impacts that make it difficult to isolate response patterns in relation to the specific logging disturbances. Methodological issues have also hampered efforts to identify generalities and achieve consensus among scientists regarding the impacts of logging on apes (Plumptre and Grieser Johns, 2001).

Three main considerations will determine how seriously wildlife populations are impacted by logging operations. First, that populations are able to survive the logging process itself, second that they are able to survive and reproduce successfully on the resources remaining after logging, and third that recolonization and population stabilization post-logging are possible (Grieser Johns and Grieser Johns, 1995). Assessment is limited by the fact that there are very few studies of change in populations from pre-logging through the logging process to regeneration. A common approach has been to compare logged and unlogged sites, and while we draw information from these studies, it should be noted that results may be confounded due to lack of information on original pre-logging population densities, which may be variable over even small areas.

Further, a temporal effect can be seen whereby patterns in responses observed immediately following logging may change as time passes. A study in East Kalimantan, Indonesia, showed that, after an initial decline related to the disturbance of the logging process, primates in general seem to cope relatively well, particularly those with a generalist diet, although it should be noted that these changes are confounded in the face of hunting (Meijaard *et al.*, 2005). Clark *et al.* (2009) sought to tease apart the direct and indirect impacts of logging on the abundance of a suite of species in northern Congo. They reported a pattern similar to that noted by Meijaard and colleagues, in that many species increased in abundance after the initial disturbance of logging had passed, linked perhaps to the opening up of the canopy stimulating new growth, and numbers returning to previous levels with time.

Although many primates are relatively tolerant of habitat disturbances, others are negatively affected, and different species may be variably impacted at a single site (Johns and Skorupa, 1987; Weisenseel, Chapman,

and Chapman, 1993; Plumptre and Reynolds, 1994; Chapman and Lambert, 2000; Paciulli, 2004; Stickler, 2004). Logging is likely to change both the abundance and the distribution of food sources in the apes' home ranges, which in turn will impact feeding strategies. These changes will alter the efficiency of foraging, which will be reflected through changes in activity budgets, the way the animal spends its time foraging, moving or resting on a daily, seasonal or other basis. For example, primates may have to forage more intensively in logged over forests to find resources (Johns, 1986b) or, alternatively, primates may adopt an energy conservation strategy, limiting activity as a result of lowered energy budgets brought about by lower resource abundance. This has been found in orangutans in monocultural acacia plantations where they feed on low-quality bark and rest much more than orangutans in natural forest (S. Spehar, unpublished data). Such effects can be identified through changes in daily ranging distance and amount of time spent feeding versus other activities.

While early studies suggested that frugivorous species are most likely to be negatively impacted by logging (Johns and Skorupa, 1987), which is particularly important given the generally frugivorous nature of the apes, a simple relationship between fruit abundance and ape persistence is unlikely to be found in most instances. For example, a meta-study of nine primate species (not including any member of the Hylobatidae) found that there was only a weak and inconsistent correlation between mortality and resource abundance and that, contrary to the results of Johns and Skorupa (1987), this was more pronounced in folivores than in frugivores (Gogarten *et al.*, 2012). This lack of a simple relationship between mortality and resource availability is likely to be because mortality is regulated by many factors, including resource abundance, disease, parasitism, and stress-related reduction in immune function. All of

“Although many primates are relatively tolerant of habitat disturbances, others are negatively affected, and different species may be variably impacted at a single site.”

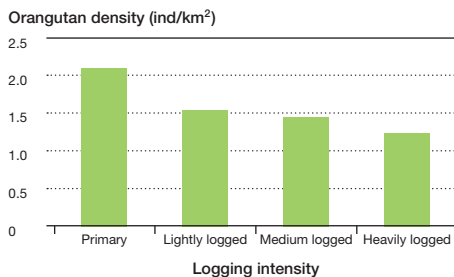
these factors act synergistically to impact animal abundance (Chapman, Lawes, and Eeley, 2006; Gogarten *et al.*, 2012), further confounding attempts to draw hard and fast conclusions about the impacts of resource extraction.

In the following sections, we summarize what is known to date of the impacts of logging on orangutans, African apes, and gibbons, and the possible mechanisms driving any changes in population density and persistence. We highlight some information gaps and provide recommendations based on this assessment.

Logging and orangutans

In Borneo and Sumatra, damage from timber harvesting is generally severe, with up to 80% damage to the canopy and potentially large ecological impacts on the apes living in these forests (Husson *et al.*, 2009; Ancrenaz *et al.*, 2010; Hardus *et al.*, 2012). Studies on Borneo show that over-harvesting of timber significantly degrades orangutan habitat and results in reduced population densities (Husson *et al.*, 2009; Ancrenaz *et al.*, 2010), and that the higher the intensity of logging, the greater the decrease in orangutan density (see Figure 3.1). Nonetheless, orangutans can survive in logged areas (Felton *et al.*, 2003; Knop, Ward, and Wich, 2004; Husson

FIGURE 3.1
Orangutan densities for Borneo under different logging intensities



Based on Husson *et al.*, 2009.

et al., 2009) and orangutan densities can be maintained with appropriate management (Marshall *et al.*, 2006; Ancrenaz *et al.*, 2010). In fact, Ancrenaz and colleagues found higher nest densities in logged forests than in nearby primary forests.

A recent large-scale nest survey on Sumatra (S.A. Wich, unpublished data) shows similar results to those from Borneo, with transects in primary forests having a higher mean number of nests per kilometer than transects in forests that have been logged. The effects of logging intensities and duration after logging could not be quantified, but, in several cases, the transects were in concessions where logging had ceased more than 20 years previously, indicating that orangutans are able to survive in such areas in the long term (Knop *et al.*, 2004). However, surveys carried out in the late 1990s recorded some transects that had been heavily logged and did not contain any orangutan nests, while adjacent primary forests still contained orangutans (S.A. Wich, unpublished data). It is difficult to be certain, but it appears that after heavy logging Sumatran orangutans disappear from logged areas. Although speculative, observations indicate that some males might move away, but that females remain and would die if food availability decreased to a level that cannot support them anymore (van Schaik, 2004; S. Wich, personal observation, 2013).

Although there is now a reasonable amount of data on changes in orangutan density associated with logging, there are fewer data on behavioral change. Some studies have assessed activity budgets during and after logging. Rao and van Schaik (1997) showed that there were more feeding bouts on leaves in the logged forest than in the primary forest. More time was spent feeding on fruits in primary than in logged forest. Both studies also showed a difference in locomotion styles between logged and unlogged, indicating that in logged forest

more energetically expensive styles of locomotion are used. More recently, a long-term study of the effects of logging on Sumatra orangutan behavior showed that orangutans spend more time travelling and less time resting in logged than in primary forest (Hardus *et al.*, 2012). Such energetically more expensive locomotion in combination with less time spent feeding on fruits could potentially lead to negative energy balances in orangutans living in logged forest, as described for the fruit-scarce periods between mast-fruited periods in primary forest at Gunung Palung in Borneo (Knott, 1998a). There is some evidence that orangutans are traveling on the ground more frequently in logged forests, thus potentially addressing those energy imbalances (e.g. Loken *et al.*, 2013). However, a follow-up study for all of Borneo indicated that although the degree of forest disturbance and canopy gap size had an influence on terrestriality, orangutans were recorded on the ground as frequently in primary forests as in heavily degraded habitats (M. Ancrenaz, unpublished data).

No other studies have been able to make such direct comparisons of behavior in logged and unlogged forest, but an alternative approach is to do a cross-site comparison and assess whether differences in activity budgets and diet between logged and unlogged sites exist. It appears that activity patterns do not show clear differences between logged and unlogged sites (see Figure 3.2); however, this rough comparison does not take into consideration potential age or sex differences, subspecies variation, or whether the sites were in dryland forests, peat swamp areas, or a mixture of the two. Nor does comparing diet across sites reveal clear differences between logged and unlogged sites (see Table 3.1), but, again, caution should be taken when comparing mean and range data without carefully controlling for the above-mentioned confounding variables. Nonetheless, both activity and diet in these



logged areas seem to be comparable to the patterns seen in orangutans in primary forests. It is also worth mentioning that the sites labeled as unlogged in the cross-site comparison have been logged since those studies took place. Consequently, Ketambe, Suaq Balimbing, Gunung Palung, Mentoko, and Ulu Segama are now sites that have undergone logging at various intensities and where data were collected when the forest was still primary. Thus, in the coming years we can expect behavioral data to come out of these sites that will allow for pre- and post-logging comparisons.

Photo: Orangutans spend more time travelling and less time resting in logged forest, which could potentially lead to a negative energy balance.
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TABLE 3.1**Orangutan diets in logged and unlogged forests**

Site and range	Fruits	Flowers	Leaves	Bark	Invertebrates	Other
Suaq Balimbing (S)						
mean	66.2	–	15.5	1.1	13.4	3.8 (inc. flowers)
<i>low fruit–high fruit</i>	62.7–69.6	–	18.3–12.7	0.8–1.4	14.6–12.2	3.6–4.1
Ketambe (S)						
mean	67.5	3.5	16.4	2.7	8.8	1.3
<i>monthly range</i>	57.5–71.5	–	10.6–20.1	2.2–3.3	5.7–11.7	–
Batang Toru (S)						
mean	73.7	5.3	6.8	2.9	2.9	8.4
Sabangau (B-L)						
mean	73.8	9.0	5.1	1.5	8.6	2.0
<i>monthly range</i>	24.4–91.9	0.0–60.2	0.3–17.4	0.0–9.1	0.7–28.0	0.1–4.9
Tuanan (B-L)						
mean	68.6	5.9	17.2	1.0	6.3	0.6
<i>monthly range</i>	26.3–88.0	0.0–5.1	4.5–49.5	0.0–5.9	0.3–24.1	0.0–2.5
Tanjung Puting (B)						
mean	60.9	3.9	14.7	11.4	4.3	4.0
<i>monthly range</i>	16.4–96.1	0.0–41.1	0.0–39.6	0.0–47.2	0.0–27.2	0.0–21
Gunung Palung (B)						
mean	70.0	5.1	13.4	4.9	3.7	2.9
<i>monthly range</i>	25.8–99.0	0.0–49.6	0.1–41.1	0.0–30.9	0.0–14.0	0.0–9.2
Kinabatangan (B-L)						
mean	68.0	1.3	22.9	6.7	1.2	–
Mentoko (B)						
mean	53.8	–	29.0	14.2	0.8	2.2 (inc. flowers)
<i>monthly range</i>	25.7–89.0	–	5.3–55.6	0.0–66.6	0.0–11.1	0.0–2.5
Ulu Segama (B)						
mean	51.5	–	35.6 (inc. flowers)	11.2	2.1	–
<i>monthly range</i>	10.0–90.0	–	8.3–75.0	0.0–36.7	0.0–8.3	–

Note: Mean values and ranges are presented. For Suaq Balimbing monthly ranges were not available, but low and high fruit availability values were available so these are reported. For Batang Toru, the “other” category includes pith and stem. Due to the preliminary nature of the Batang Toru data monthly ranges are not yet known. Data were not available from some sites for some food items. S = Sumatra, B = Borneo, L = logged. Based on Morrogh-Bernard *et al.* (2009) and Wich *et al.* (2013).

If forests are allowed to regenerate, the longer-term impacts of unsustainable logging can be limited as long as the logged area is adjacent to forest where orangutans still exist. Recolonization can even occur in cases where the logging intensity was at such a level that it led to the complete disappearance of orangutans (e.g. Knop *et al.*, 2004). With time, orangutan populations are able to recover to pre-logging densities if the volume of timber harvested was low and residual forest damage was limited. However, in Southeast Asia, the level of damage that occurs during the logging process is usually significant and, as a result, orangutan densities tend to be much lower.

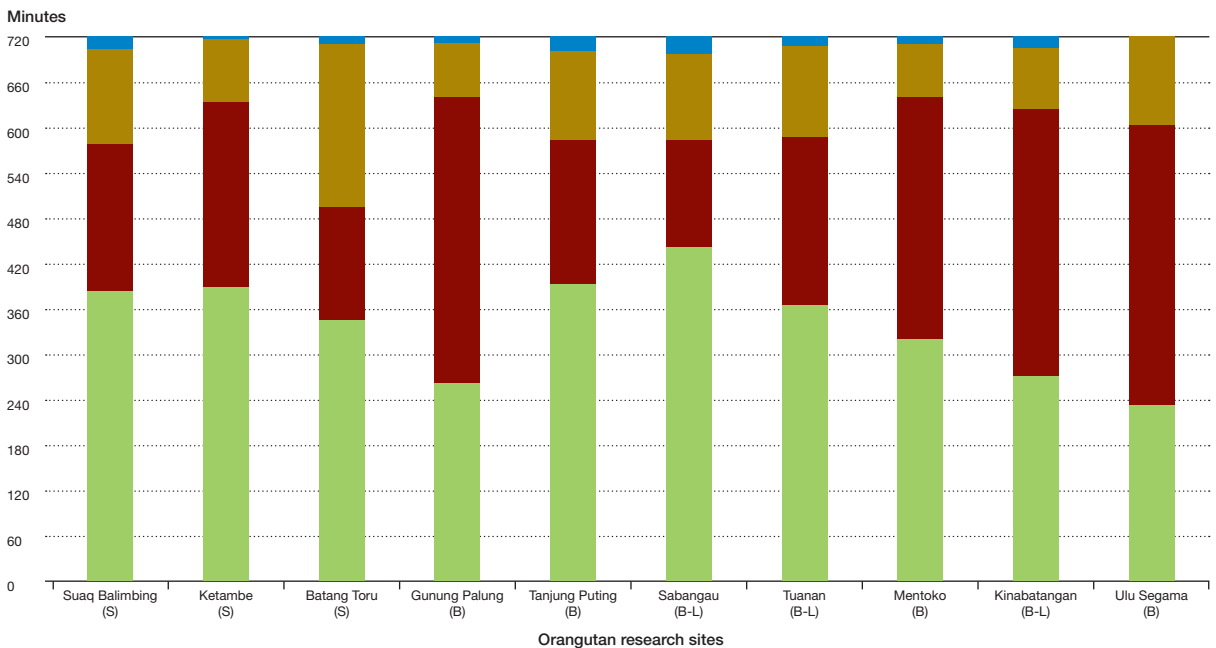
Overall, the findings of recent studies indicate that conventional logging practices will cause decreases in orangutan density (but see Marshall *et al.*, 2006), although these decreases are likely to become less marked

as the forests have time to regenerate and densities gradually increase again through recovery or recolonization. In addition, conventional logging seems to have no large effects on activity budgets and diet once logging has ceased. Both these findings argue that logging concessions have an important potential role in orangutan conservation as long as they are well managed in regards to both their direct and indirect impacts, where, for the latter, the control of hunting and poaching is vital (Meijaard *et al.*, 2012; Chapter 6). Concessions where RIL (as opposed to conventional) practices have been used tend to have higher orangutan densities (Ancrenaz *et al.*, 2005, 2010). For the survival of orangutans, it is therefore not of crucial importance whether or not logging occurs, but whether this logging uses reduced impact methods and how much time a forest is given to recover following logging.

FIGURE 3.2

Orangutan activity budgets in a 12-hour day

Key: ● Feed ● Rest ● Travel ● Other
S = Sumatra, B = Borneo, L = logged



Based on Morrogh-Bernard *et al.*, 2009; Wich *et al.*, 2012.

Logging and African apes

Studies of the African apes in logged forests thus far have produced ambiguous results and have failed to identify consistent patterns of impact. Whilst conventional logging has definite negative impacts on ape populations (Morgan and Sanz, 2007), the impacts of selective logging are less clear. Bonobos have not been studied in logging concessions, whereas some gorilla and chimpanzee populations in logging concessions have been monitored for more than a decade. Some nest-count surveys have indicated that gorillas are relatively unaffected by logging once the initial disturbance has passed (White and Tutin, 2001; Arnhem *et al.*, 2008) and, indeed, longer-term studies have found gorillas occurring at fairly high densities in concessions in northern Congo that are considered to be well-managed (Morgan and Sanz, 2006; Stokes *et al.*, 2010). Nonetheless, gorilla densities decline in proximity to the roads and human settlements throughout logging concessions (Poulsen, Clark, and Bolker, 2011; see also Chapter 6), indicating possible variability in population responses within active or previously logged concessions.

For chimpanzees, the picture is less clear; an early investigation in Uganda demonstrated an inverse relationship between logging intensity and chimpanzee density, and identified the degree of habitat disturbance as a key factor in determining chimpanzee abundance in post-logged forests (Skorupa, 1988). Subsequent nest-count surveys at various sites found no consistent response: some chimpanzee populations decreased, others increased or showed no change (Plumptre and Reynolds, 1994; Hashimoto, 1995; White and Tutin, 2001; Dupain *et al.*, 2004; Matthews and Matthews, 2004; Arnhem *et al.*, 2008). The accuracy of nest counts can differ, depending on survey intensity and ability to assess nest decay rates. However, long-term monitoring of chimpanzees in logged and unlogged habitats in

northern Congo has been able to detect preferences for less disturbed forest and suggests that chimpanzees are more adapted to mature forest (Stokes *et al.*, 2010; D. Morgan, C. Sanz, S. Strindberg, J. Onononga, C. Eyana-Ayina, and E. Londsorf, personal communication, 2013). Even if they avoid human contact and favor mixed mature forest for nesting, chimpanzees seem to be able to slowly restore a stable population in regenerating forest on logging concessions if hunting pressure is controlled. Over the long term, chimpanzee densities in forests logged 15 years prior remained low compared to unlogged habitat in Congo (Stokes *et al.*, 2010). Similarly, a 28-year study of primates in Uganda has shown that chimpanzees consistently occur at lower densities in logged areas than in unlogged areas (Chapman and Lambert, 2000).

Apes generally move away from operational areas and their forced migration into adjacent home ranges will stress both immigrant and resident apes. It has been suggested that, in the short term at least, chimpanzees appear to be more negatively impacted than gorillas by the disturbance associated with logging (e.g. Arnhem *et al.*, 2008). A plausible explanation for this is that chimpanzees are territorial and incursions into another chimpanzee community's home range are generally hostile (Mitani, Watts, and Amsler, 2010). Logging activities will displace resident chimpanzees and may force them to encroach on a neighboring community's home range, resulting in social upheaval and sometimes in lethal conflict: females might be able to transfer between groups, but males are likely to be attacked and possibly killed. Aggressive intercommunity interactions in association with logging are thought to have reduced chimpanzee densities at Lopé in Gabon (White and Tutin, 2001). Gorillas are not territorial and it has been suggested that they do not have the same constraints on their movements as chimpanzees, and this may help them to resist the impacts of forestry

“Logging activities will displace resident chimpanzees and may force them to encroach on a neighboring community's home range, resulting in social upheaval and sometimes in lethal conflict.”

activities. However, the vulnerability of gorilla group stability should not be overlooked: extreme social disruption leads to higher rates of infanticide in mountain gorillas (Kalpers *et al.*, 2003).

To date, there has been little research on how changes in forest productivity may ultimately affect the demography and density of ape populations. However, rare insights into the impacts of logging on chimpanzee ecology and reproductive fitness come from ongoing studies at Kibale in Uganda, where logging took place in the 1960s, with the intensity of timber extraction varying between logging compartments. Female chimpanzees had lower reproductive success with longer interbirth intervals and higher infant mortality in areas with outtake rates of 17.0 m³/ha (50.3% of basal area reduction) and 20.9 m³/ha (46.6% basal area reduction) than females residing in less disturbed forests (Emery Thompson *et al.*, 2007). One might conclude that more intensive logging regimes had reduced the food resource base for chimpanzees. However, more recent research indicates that the explanation may be more complex because the impact of logging on the chimpanzees' diet was low, even in cases where preferred food items had been exploited (Potts, 2011). In Potts' study, chimpanzee abundance did not appear to be related to logging history, highlighting the fact that previously logged forests may still retain resource attributes important for ape survival. However, it is important to consider the difference in spatial and temporal scales of these investigations and that indirect impacts could also be influencing chimpanzee densities (see Chapter 7).

The density data compiled in Annex II show that both chimpanzees and gorillas are able to persist in timber production forests, but with varying degrees of success and undetermined prospects for long-term survival. Studies in northern Congo indicate that Forest Stewardship Council (FSC) cer-

tification processes have positively benefitted conservation in the context of timber exploitation (Stokes *et al.*, 2010; Morgan *et al.*, 2013); however, it has not yet been determined if and how specific low-impact logging practices are affecting gorillas and chimpanzees. See also the Goulougo Triangle case study and the Wildlife Wood Project (WWP) case study in Chapter 4 (pages 117 and 120).

Logging and gibbons

As with the great apes, the impacts of logging on gibbons are somewhat equivocal. There are doubtless numerous variables which interact to determine how well gibbons are able to persist and recover after logging. These variables include the intensity and extent of logging operations; the incidental damage incurred to habitats during operations; the time since the logging event; the silvicultural techniques used before, during, and after logging; the species of tree targeted for extraction, and the resident population's reliance on them as keystone species or fall-back resources; the taxon's dietary flexibility; how marginal the site was for gibbon persistence pre-logging; degree of competition with sympatric taxa; and the severity of any additional anthropogenic impacts such as hunting, road access, human influx, and agricultural expansion. It is therefore not surprising that clear themes on the impacts of logging on gibbon persistence are difficult to isolate, especially given the family's large geographic range.

Within the Hylobatid family, the genus *Hylobates* is the best studied in terms of impacts of logging on population densities. The most comprehensive study to date was conducted on *Hylobates lar* in Peninsula Malaysia and tracked gibbon density prior to logging through the logging process and followed up post logging, spanning a research period of over 12 years. Johns and colleagues (Johns, 1986b, 1992; Grieser Johns

and Grieser Johns, 1995) found that there were no clear trends in density of gibbons at the site over this period, including no sign that the population had decreased post logging, despite increased mortality during the logging process itself. Conversely, Southwick and Cadigan (1972) found in their study of *H. lar* that group densities were marginally higher in primary forest (0.43 groups per km²) than disturbed or secondary forests (0.34 groups per km²) caused by selective logging in the past. Pileated gibbon (*Hylobates pileatus*) in Thailand have lower densities and tend to avoid selectively logged areas and even areas of undisturbed forest nearby (Brockelman *et al.*, 1977). Gibbon densities in areas which had not been logged since the 1970s were almost three times higher than those logged in the 1990s but still lower than those in pristine conditions, suggesting some recovery over long time periods but probably restricted by lower resource abundance (Brockelman and Srikosamatara, 1993; Phoonjampa *et al.*, 2011).

Studies of Müller's gibbon (*Hylobates muelleri*) on Borneo are contradictory. One study showed no difference in group density between primary forest and low intensity, selectively logged forest (Wilson and Wilson, 1975). A second study showed decreases in group density from 7.3 groups per km² in primary forest, 5.0 groups per km² in forest logged three to five years previously, and 2.3 groups per km² in forest logged one week previously, suggesting populations go through a bottleneck caused by mortality, or possibly migration out of the area, at the time of logging with subsequent recovery still not complete five years later (Wilson and Johns, 1982). Another Bornean gibbon species, the Bornean white-bearded gibbon (*Hylobates albibarbis*), living in peat-swamp forest in the Sabangau catchment, Central Kalimantan, Borneo, has been shown to have densities correlated with canopy cover and

tree height and it has been surmised that, at one site, 30 years of logging had negatively impacted gibbon densities (Buckley, Nekaris, and Husson, 2006; Hamard, Cheyne, and Nijman, 2010). Conversely, a study on Kloss's gibbon (*Hylobates klossii*), a species endemic to the Mentawai Islands, Indonesia, showed no difference in densities between unlogged forests and those logged 10–12 years and 20–23 years earlier (Paciulli, 2004). Paciulli (2004) surmised that this lack of relationship between density and logging may be because tree species targeted by loggers are dipterocarps, which are not used by *H. klossii* as a feeding resource (Whitten, 1982), suggesting that the resource base was not impacted by the logging regime. However, this hypothesis disregards the likely significant incidental damage caused by the logging process.

Information from the other three genera of the Hylobatidae is generally lacking, being largely comprised of anecdotal observations. For example, the siamang (*Symphalangus syndactylus*) reportedly occurs in lower densities in logged over forests in southern Sumatra (Geissmann, Nijman, and Dallmann, 2006), an observation apparently borne out by lower recorded densities in forest disturbed by logging (0.20 groups per km²) compared to undisturbed habitats (0.42 groups per km²) (Southwick and Cadigan, 1972). Qualitative observations suggested that the northern yellow-cheeked gibbon (*Nomascus annamensis*) was absent in several areas that had been subjected to logging in southern Laos (Duckworth *et al.*, 1995; Evans *et al.*, 1996). However, high hunting pressure may have confounded these assessments (Duckworth *et al.*, 1995) as they probably do for all *Nomascus* species (Duckworth, 2008; Rawson *et al.*, 2011). Large home range size in the eastern black-crested gibbon (*Nomascus nasutus*) was anecdotally attributed to forest degradation caused by logging, specifically loss of fruit trees.

Where detected, changes in population density may be driven by a number of factors including direct and indirect mortality, changes in resource abundance and habitat fragmentation. Gibbons, due to their territorial and strictly arboreal nature, may be more affected by the immediate impacts of logging regimes than many other wildlife species. Gibbons have been shown to stay in their home ranges during logging activities because of their territoriality, maintaining distance from areas actively being logged by staying in unlogged or already logged areas within their home ranges, and only marginally travelling outside the home range to skirt logging activities if necessary (Wilson and Johns, 1982; Johns, 1986b). It is surmised that in instances where gibbons are forced from their home ranges during logging operations, high levels of mortality will result (Johns and Skorupa, 1987), with constant displacement by resident gibbon groups, unfamiliarity with the distribution of resources, and stress all playing a role. Additionally, their arboreal nature coupled with fragmentation of home ranges by logging roads and tree falls may also limit their ability to effectively avoid logging operations (Meijaard *et al.*, 2005) and may also result in increases in fatal falls. These factors may result in the complete loss of groups from areas during the logging process (e.g. Fan Peng-Fei, Jiang Xue-Long, and Tian Chang-Cheng, 2009).

Increases in infant mortality for resident gibbons may also occur during the logging process. Infant mortality in all primates commonly increases at times of environmental stress and resource shortages (Dittus, 1982; Hamilton, 1985; Gould, Sussman, and Sauther, 1999), and pregnancy and lactation are particularly energetically expensive for female mammals (Clutton-Brock, Albon, and Guinness, 1989; Rogowitz, 1996; Lee, 1998). Displacement and stress caused by logging, plus changes in the abundance and

distribution of resources within a home range may negatively impact females' energy budgets, with subsequent nutritional impacts on dependent infants. Significantly, Johns (1986a) found that, when subjected to selective removal of timber, infant mortality in

Photo: Gibbons, due to their territorial and strictly arboreal nature, may be more affected by the immediate impacts of logging regimes than many other wildlife species. © Terry Whittaker



“It is quite common for extraction crews to be involved in hunting during operations and the volumes of bushmeat consumed can be staggeringly large.”

a population of *H. lar* was 100%. Although the cause of this was not described, it was likely through abandonment and infant malnutrition (Meijaard *et al.*, 2005).

Finally, an indirect impact of the logging operation itself on gibbons can be increased levels of hunting (Bennett and Gumal, 2001; see also Chapter 7). It is quite common for logging crews, for example, to be involved in hunting activities during operations and some reports suggest the volumes of bushmeat consumed can be staggeringly large, for example, 29 086 kg, including 445.5 kg of primates, in one year for one logging camp in Sarawak (Bennett and Gumal, 2001). For hunters using guns to take species such as deer and bearded pigs, gibbons can make relatively easy targets, particularly because of gibbons' proclivity to vocalize loudly in the morning from fixed locations (Bennett and Gumal, 2001). Areas with high hunting pressures may have localized extirpation of gibbon populations (Duckworth, 2008; Rawson *et al.*, 2011), and even low levels of off-take can impact population viability in already small and vulnerable populations (e.g. Waldrop *et al.*, 2011). As such, control of hunting, specifically with guns, during logging may be an important determinant of gibbon persistence and recovery.

While it appears clear that increases in mortality occur during the logging process, as described above, the ability of gibbons to adapt to and recover in forests post logging is less conclusive. Johns and Skorupa's (1987) review of the literature relating to impacts of logging on primates showed that a primate species' degree of frugivory was negatively correlated to persistence in recently logged forests, in contrast to more recent meta-studies (Gogarten *et al.*, 2012). This relationship is especially relevant for gibbons given their large reliance on fruit sources both as primary food sources and fallback resources (Bartlett, 2007). Some commentators have maintained that selective logging

will have little effect on gibbon populations as gibbon diets are relatively flexible so the removal of food trees either deliberately or incidentally will only change relative species utilization in the diet (Chivers, 1972; Wilson and Wilson, 1975). Gibbon responses to this relatively quick change in the availability of food resources, specifically fruit, will likely depend on behavioral and dietary flexibility, including the ability to rely on low quality leafy matter. Gibbons possess simple stomachs, thus do not have the same ability to digest foliage as the often sympatric colobine monkeys, such as leaf-monkeys or langurs (e.g. *Trachypithecus* and *Presbytis* species), which possess specialized stomachs and symbiotic bacteria which break down and aid in digestion of leaf cellulose (Raemaekers, 1978; Chivers and Hladik, 1980; Chivers, 1994; Caton, 1999). As fruits generally have more free sugars available than leafy matter (Raemaekers, 1978; Johns, 1986b) this may in turn impact energy budgets and, potentially, mortality and fertility.

Existing evidence suggests that gibbons are likely to change behavior in response to changes in resource availability brought about by logging events. Gibbons commonly reduce ranging behavior and other activities in times of low resource abundance under natural conditions, for example when fruit is not seasonally available (Chivers, 1974; Raemaekers, 1980; Gittins, 1982; Fan Peng-Fei and Jiang Xue-Long, 2008). In his comparison between pre- and post-logged forest, Johns (1986b) found that gibbons responded similarly, with significant reductions in activity levels post logging. These changes in activity patterns in response to changes in resource abundance may be functional, however if there are insufficient resources, this may result in negative energy budgets, resulting in increases in mortality through starvation and associated factors. Lowered energy budgets will have different impacts on different age and sex classes in gibbons.

During pregnancy and lactation, adult females have considerably higher metabolic requirements per unit body weight, as do juveniles due to growth trajectories. Juveniles are also less efficient foragers and may suffer displacement from preferred food resources (e.g. Fan Peng-Fei and Jiang Xue-Long, 2010). Thus, under conditions of sub-optimal food availability, we may predict increased mortality in juveniles and infants (O'Brien *et al.*, 2003; Meijaard *et al.*, 2005; Rawson, 2012). This may also result in declines in birth rates and/or infant survival as females may not be able to maintain pregnancy or lactation on a low energy diet; both possible outcomes would impact the demographic structure of the population.

One study on gibbons does bear out a direct link between lowered resource abundance and increased mortality in infants and juveniles that may be applicable to logging scenarios. O'Brien *et al.* (2003) studied siamang in forest areas subjected to severe fires in 1997 and compared them to those in forests which did not experience fires. Areas subjected to fire suffered mortality of 25% of trees including the loss of almost half of the population of strangling figs, a key siamang resource, followed by ongoing high levels of tree mortality. Infant and juvenile mortality in groups living in fire-impacted areas was significantly higher, with 30% fewer infants, 24% fewer small juveniles, and 39% fewer large juveniles. After several years, groups in fire-impacted areas had declined in number compared to control groups. The impacts on infant and juvenile survival, leading to changes in the demographic structure of the population, were attributed to a reduction in the availability of food resources. This was brought about by tree mortality and may therefore provide some proxy for initial impacts in a logging scenario.

Here we suggest that while responses by gibbons to logging operations will not be uniform, there is potential for them to impact

long-term viability of resident populations. Increased levels of mortality, especially amongst infants and juveniles, appear to be likely, which may have long-term impacts on the demography and therefore viability of the population. Populations which are already suppressed due to hunting are likely to be particularly vulnerable due to gibbons' slow reproductive rates. We also suggest that the dietary flexibility of gibbons in response to logging events may not always be sufficient to overcome impacts on energy budgets, and increased mortality, again, especially in infants and juveniles, and lowered fertility may also result in some circumstances. Comparative ecology also suggests that some gibbon taxa may be more affected by changes in resource abundance than others. For example, it has been noted that siamang (*Symphalangus syndactylus*) densities may be reduced less by logging than the densities of sympatric agile gibbons (*H. agilis*) owing to the former's naturally more folivorous diet (Geissmann *et al.*, 2006). Additional longitudinal studies following a population from pristine to post-logged forest are likely needed to tease out the full impacts on resident gibbon populations.

As discussed above, recovery of gibbon populations post logging is likely to be linked to the impacts of logging on key-stone food resources and the demographic profile of the populations, particularly where populations are already suppressed. In addition, changes in forest structure caused by selective logging practices and infrastructure for timber removal are likely to impact resident gibbon populations after the logging teams leave. Logging and associated infrastructure may cause habitat fragmentation, where a formerly contiguous area of forest becomes discontinuous sections (see Chapter 7 for more information on habitat fragmentation). Under these circumstances demographic variability, natural stochastic events such as disease and natural disaster, inbreeding depression, as well

“Recovery of gibbon populations post logging is likely to be linked to the impacts of logging on key-stone food resources and the demographic profile of the populations.”

as anthropogenic influences may make small populations in forest fragments more susceptible to localized extinction than those in larger areas with larger populations (Fahrig and Merriam, 1994).

As gibbons can become isolated by even small openings in the canopy (Johns, 1986b; Choudhury, 1990; Sheeran, 1995), fragmentation must be considered a potentially significant issue. The isolation of populations from one another may lead to prevention or retarding of gene flow between populations. Recolonization of fragments where local extirpations have occurred, which may be vital for species conservation at the landscape level (Fahrig and Merriam, 1994), will also be problematic in highly fragmented landscapes. At a more local level, isolation may also impact dispersal of gibbons. Gibbons generally leave their natal territory upon reaching maturity to form their own group; however, fragmentation may prevent this dispersal (Kakati *et al.*, 2009). Despite their acrobatic nature and apparent comfort brachiating through a complex three-dimensional environment, gibbons are subject to high levels of injury, and presumably mortality, through falls. Schultz (1939) found that 36% of gibbons in his sample of 118 wild caught individuals had long-bone fractures (some more than one) which had subsequently healed and were likely attributable to falls (Gibbons and Lockwood, 1982). It is logical that incidences of falls may be exacerbated by reduced availability of supports for arboreal travel, increased canopy gaps and the unfamiliarity of routes associated with habitat fragmentation caused by logging. One solution which has been successfully tested for gibbons is the construction of canopy bridges which may reduce incidences of falls and the need for terrestrial travel (Das *et al.*, 2009).

The minimum fragment size for maintaining gibbon populations has been assessed in two taxa with similar results. Gray *et al.* (2010) modeled minimum fragment size of

evergreen forest for southern yellow-cheeked gibbon (*Nomascus gabriellae*) persistence in a naturally fragmented landscape in Cambodia, finding that areas > 15 km² were required to maintain a viable population. Kakati *et al.*'s (2009) assessment of western hoolock (*Hoolock hoolock*) in a fragmented landscape in India suggested that populations in areas < 5 km² had smaller group sizes and higher mortality and were more likely to suffer localized extirpation than those in larger fragments > 20 km². This suggests that fragmentation of habitat, when severe enough to reduce forest patches to < 20 km², may be highly detrimental to the long-term persistence of gibbon populations. Yanuar and Chivers' (2010) study in five sites in Indonesia suggests that for the agile gibbon (*Hylobates agilis*) and the siamang (*S. syndactylus*), fragmentation of the forest leads to behavioral changes, such as reduction in home range size and change in diet due to changes in forest composition, which may also impact the long-term viability of these groups.

Only one case study of the impacts of logging on forest fragmentation and persistence in gibbons is available, that of the eastern hoolock (*Hoolock leuconedys*) in China, which has been heavily impacted by commercial-scale logging. Road networks and extraction of timber have resulted in severe fragmentation of gibbon habitat, with the total population now residing in 17 fragments and none having more than five groups (Fan Peng-Fei *et al.*, 2011b). A 50% decline occurred between 1994 and 2009 in five sites and extirpation in nine sites has been recorded, including the country's previously largest population (Fan Peng-Fei and Huai-Sen Ai, 2011; Fan Peng-Fei *et al.*, 2011b). Logging operations and effects from fragmentation are thought to be significant factors, although hunting has played a confounding role.

While demographic composition and general population health may return to pre-

logging levels despite increases in mortality at the time of logging, populations which are already heavily impacted by other processes, such as hunting or habitat fragmentation, may not recover. Gibbons have long interbirth intervals and late sexual maturity resulting in low lifetime reproductive output (Palombit, 1995; Bartlett, 2007; Reichard and Barelli, 2008) and as such, even small increases in mortality in small populations may lead to loss of population viability (Waldrop *et al.*, 2011). Logging in areas with small vulnerable gibbon populations, especially those taxa that are highly globally threatened and/or range restricted, should therefore be conducted only with considerable assessment of the potential impacts.

Studies of the direct impacts of mining on ape populations

Mineral and hydrocarbon developments result in broad-scale changes to habitat structure and composition as a direct result of activities during the different phases of mining, oil, and gas projects (see Chapter 5 for more information on these phases). Seismic surveying and exploratory drilling require the clearing or disruption of only a few hectares of vegetation in each site, but there could easily be hundreds of such sites scattered across the landscape, and infrastructure development will fragment the habitat. Further, noise associated with seismic surveys has been shown to displace wildlife (Rabanal *et al.*, 2010). Displacement and disturbance also occur as the number of people in the forest increases during exploratory operations (Chapter 7).

The implementation phase of a project typically results in the most dramatic ecological changes and greatest period of disturbance for biodiversity in general. Implementation activities may include more complete devel-

opment of a transportation network; construction of drilling and extraction sites; and construction of facilities. The operation phase generally results in continuous day-to-day production; maintenance of facilities; and transport of the extracted materials via pipelines and export terminals. Although the ultimate impacts of these activities on biodiversity are often similar, they may differ in source, area affected, scale, intensity, and boundaries of responsibility.

The study of the impacts of extractive industry on wildlife is still nascent and is yet to provide a detailed picture of the consequences of mining operations or of the cumulative impacts that may occur. Research is needed to assess the impacts of each phase of project development, both in mine site areas and along key sections of the transport corridor. However, the above observations suggest that the risks and threats to apes are potentially very high over the life of a resource extraction project, and severe negative impacts may occur, increasing in intensity unless appropriate impact avoidance, minimization, and compensation measures are implemented early in a project's life.

“The study of the impacts of mining on wildlife is still nascent and yet to provide a detailed picture of the consequences of mining operations.”

Mining and orangutans

The impacts of mining activities on orangutans have been studied in much less detail than those associated with timber extraction. Although no comprehensive studies have been conducted on the impact of mining on orangutans, it is obvious that the mining industry is a potential threat to orangutan habitat in a number of important areas. Anecdotal information and observations suggest that where open-pit mining and orangutan habitat overlap, orangutans are generally ignored, but some are translocated (relocated) during mine development, with likely detrimental outcomes for the orangutans. This is primarily a concern where coal and bauxite deposits significantly overlap



Photo: The establishment of mine sites, roads, and associated infrastructure in natural forest has a direct impact on orangutans and other biodiversity.
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with orangutan habitat and open-pit mining is practiced.

Mining concessions often cover large areas of prime orangutan habitat. The establishment of mine sites, roads, and associated infrastructure in natural forest has a direct impact on orangutans and other biodiversity. There are no scientific publications in the peer-reviewed literature that report on the impacts of mining on orangutans. At least one company, however, reports its own findings with regard to forest and orangutan management. Kaltim Prima Coal (KPC) reported in their 2010 Sustainability Report (KPC, 2010, p. 63) that “fauna monitoring in 2010 was done to inventorize the orangutan as a protected endangered species. [...] The conclusion of this activity is that the orangutan uses vegetation resources in the mining reclamation area as its source of food and trees as nests, this is shown by the many nests and scratches in the tree trunks in the reclamation area.” This company also relocates orangutans found at their mining sites to safer locations; however, no popula-

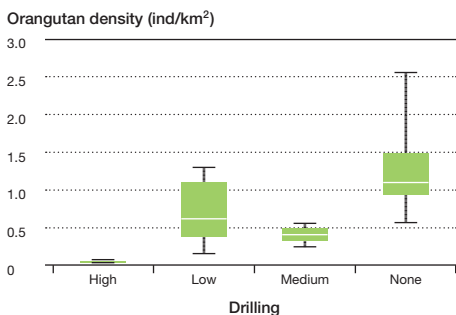
tion trends or success rates of the translocations are known.

The establishment of opencast mines and access roads generally results in clear cutting of much of the vegetation. This leaves little habitat in which orangutans can survive, nor the opportunity to successfully manage any orangutans that do survive in such areas. In many cases, the only option has been to translocate orangutans from these cleared areas to nearby forests with the help of government agencies and orangutan welfare organizations. However, translocation can create ecological problems (e.g. orangutan numbers exceeding the carrying capacity of the area into which they are moved, introduction of disease, disruption of the original social network) and only offers a partial solution to the problem of keeping orangutans out of operational areas. This suggests that large-scale mining is of most concern with regard to orangutans. However, a World Bank study in 2000 suggested that artisanal and small-scale mining (ASM) might be more harmful to the environment (McMahon *et*

al., 2000). For more information on ASM see Chapter 6.

Unfortunately, there are hardly any data on the potential impacts of the exploration phase on orangutans. The only dataset we are aware of comes from southwestern Sumatra. Here, the impact on orangutans of drilling activity during the exploration phase was assessed in the Batang Toru area. Standard line transects were conducted in this area and orangutan densities were determined for each phase. Drilling intensity for each transect was determined by assigning a drilling intensity category to each transect (ranging from none to high, based on the number of drill holes per unit area). These results show that there is a significant negative effect of drilling intensity on orangutan density (Figure 3.3). Thus, high intensity drilling negatively impacts orangutan density, whereas low and medium intensity exploration does not significantly decrease orangutan density. In this case, there were no access roads in the forest and physical damage to the forest was limited. As a result, it is likely that orangutans in this area shifted location within their home range during the drilling phase and that there were no actual long-term decreases in orangutan density.

FIGURE 3.3
Boxplot showing the orangutan density (ind/km²) for three categories of drilling intensity and one area without drilling



Adapted from S. Wich and M. Geurts in PT Newmont Horas Nauli (2003). Courtesy of S. Wich.

Orangutans are ecologically relatively versatile and can be expected to recover to some extent following high-quality land rehabilitation after mining, especially if this is done with native species that provide food to orangutans. However, they are not expected to attain the same densities in rehabilitated areas as in primary forest, partly because human disturbance is likely to be high in such former mine areas. A good example is the KPC mine in East Kalimantan, where orangutans still occur after decades of coal mining, although apparently at low densities. This mine borders Kutai National Park, which could provide a refuge. For more information on Kutai National Park and the KPC mine see the case study in Chapter 2 (page 43).

Mining and African apes

Despite numerous environmental impact assessments (EIAs), there have been very few studies of African apes at mining sites. Such studies have been implemented only relatively recently, because baseline data often do not exist or, if they do, data sharing is restricted by confidentiality clauses. Rabanal *et al.* (2010) reported that noise associated with seismic surveys in Gabon displaced gorillas and chimpanzees for many months after operations had been completed, which could result in increased inter- and intra-specific conflict as animals are forced into neighboring home ranges or feeding and nesting sites within their range are disrupted. Observational and conjectural data derived from recent field studies carried out in the vicinity of extractive industry sites provide some insight into probable risks and threats to apes during the extractive industry life cycle. Ecologically, great apes and the habitat they depend on appear to be experiencing a two-fold threat in both mining sites and transport corridors. Table 3.2 summarizes some of the potential impacts on apes for each phase of mine development.

Mining and gibbons

The extent and impacts on gibbons from mining operations are poorly understood and documented. Of the Hylobatid taxa currently listed on the IUCN Red List of Threatened Species (IUCN, 2013), mining is only mentioned as a potential threat to two species, *Hoolock hoolock* and *H. leuconedys*. A trawl of the literature on the Hylobatidae comes up similarly short. While mining is occasionally mentioned as a potential threat, information as to the extent, intensity or nature of the threat is unavailable.

For example, opencast mining and oil drilling were identified as a threat to *Hylobates* species on Indonesian Borneo, Sumatra, and Java but the threat was not quantified or qualified because information is generally lacking, and perhaps as such in a ranking exercise it was listed as priority 19 of 20 for gibbon conservation (Campbell *et al.*, 2008a). Likewise, opencast coal mining, limestone mining, and oil drilling and exploration are mentioned in the literature as having impacted western hoolock (*H. hoolock*) gibbon habitats in northern India (Choudhury, 2006, 2009), but how and to what extent is not detailed. It appears based on the evidence, or lack thereof, that either mining poses a minimal threat to gibbons relative to other threats or that the degree of threat is not yet appreciated by those engaged in gibbon conservation.

However, mining operations and gibbon distribution do co-occur in many landscapes. A recent analysis (UNEP-WCMC, 2012) found that only two Hylobatid taxa did not have mining operations within their global ranges: *Nomascus nasutus* and *N. hainanus*. This is perhaps not surprising given that these species have a global area of occurrence of only a few thousand hectares with global populations of approximately 130 and 23 individuals, respectively. However, this initial analysis (UNEP-WCMC, 2012) also found that no more than 0.02% of any

of the 16 taxa of gibbons' assessed global range fell within known areas of mining and the number of 1-km² pixels occurring within any taxon's range was under 60 in all instances. This represents a very small proportion of global ranges for most taxa (see Chapter 5 for more detail). Those species which were predicted to be most impacted by mining operations, based on (1) the overlap between mining activities and global range; (2) a large proportion of mines in what may be core areas; and (3) productive mines in the protected area network, were *H. lar* and *H. muelleri* (UNEP-WCMC, 2012).

Impacts of these extractive industries on gibbon ecology will however depend on the scale and nature of operations. Surface mining projects such as opencast mining and strip mining are, of course, highly disruptive for gibbons as the forest is clear-felled in order to remove the overburden. Given gibbons' arboreal nature and reliance on forests, surface mining and gibbon persistence are clearly incompatible (Cheyne *et al.*, 2012). Gibbons, under these circumstances, may be forced from the area, despite their territorial nature. As discussed in the logging section, this may cause high rates of mortality and is likely to create increased competition for remaining resources and the possibility of a future population reduction.

In addition to the direct impacts on habitat of mining operations themselves, associated infrastructure development including access roads and provision of power supply may have detrimental impacts on gibbons. Most significantly these may fragment the landscape, provide improved access for hunters and permit access into remote regions for in-migration and conversion of forest for agriculture (see the earlier logging section for a discussion of the implications of these impacts and Chapter 7 for more information on indirect impacts).

Anthropogenic sound generation has been shown to have potentially negative impacts on a wide range of wildlife species

“In addition to the direct impacts of mining operations, associated infrastructure development including access roads and provision of power supply may have detrimental impacts on gibbons.”

due to its ability to mask calling behaviors, induce stress, displace animals, change behavior, for example increasing vigilance activities, and distract animals, resulting in predation or a reduction in time available for other important activities (see Chan and Blumstein, 2011 for a review). This is likely to apply to gibbon groups living in association with mining operations or result in displacement from territories. For example, Delacour's langur (*Trachypithecus delacouri*) groups reportedly changed their home range in response to nearby blasting of limestone (Nguyen Vinh Thanh and Le Vu Khoi, 2006) while a range of taxa, especially wide ranging taxa, changed their behaviors in response to noise associated with oil prospecting (Rabanal *et al.*, 2010); however, this remains supposition at this time.

ASM has larger environmental impacts per unit of production than industrial-scale mining; however, these impacts are largely spatially restricted owing to their smaller operation sizes (Hentschel, Hruschka, and Priester, 2002). ASM is a recognized driver of deforestation, and may act to fragment the landscape at the local scale (Hentschel *et al.*, 2002), with impacts on gibbons as discussed under habitat fragmentation in the logging section. However, the most significant impacts on biodiversity are the result of pollution in waterways but how seriously this influences gibbon ecology is unclear, though the effects is likely to be small.

Mining impacts on the Hylobatidae, both in terms of severity and extent, represent a large information gap. It has been noted that there is very little conservation work with gibbons in mining or logging concessions in Indonesia. A barrier is the lack of engagement of the companies in conservation issues and the fact that the conservation threats to the gibbons can be overshadowed by other high profile taxa, such as the orangutans (S.M. Cheyne, personal observation, 2013). Raising the profile of gibbons as a threatened taxon, which is potentially neg-

atively impacted by mining operations and other extractive industry, may prove beneficial in addressing these information gaps and gaining an improved understanding of the relative threats, as well as approaches for mitigation.

Potential long-term impacts and future studies

The impacts of extractive industries on ape populations are likely to be severe and long lasting, but thus far few studies have been able to detect, let alone measure them beyond changes in population densities. Surveys of apes generally use proxies for the animals rather than direct observations, for example nest counts for great apes and point count vocal surveys for gibbons. Approaches commonly vary between studies, which limits comparability of the data (Kühl *et al.*, 2008). However, a fundamental issue in determining how extractive industry impacts animal density is that most studies involve comparison of population density in theoretically matched exploited and unexploited areas rather than longitudinal studies at a single site. As densities may vary naturally over small spatial scales, such approaches further confound efforts to determine the impacts of extraction on resident ape populations. Additional long-term studies which use uniform methods for determining density from pre- to post-extraction at the same site are needed to elucidate the long-term impacts of the various extractive industries on apes. New techniques, such as the ability to ascertain population size and structure by genotyping DNA extracted from feces that have been collected non-invasively (e.g. Arandjelovic *et al.*, 2011), will improve the reliability of future surveys of ape population size estimates.

Measuring specific impacts on apes is problematic for a number of reasons, and the complexity of trying to isolate specific

factors in any ecosystem is mentioned above, however, a major obstacle to behavioral observations is that apes are extremely wary and generally flee when they see, hear, or smell a human. Therefore studies of ape behavior, particularly in low visibility environments, generally require that animals become habituated to human observers. With orangutans, this process is rapid, but it can take several years with African apes (Williamson and Feistner, 2011). In addition, to calibrate change linked to extraction, habituation should be initiated before the onset of industrial activities. Such foresight led to the establishment of the Goulougo Triangle Ape Project, where researchers began to habituate and study gorillas and chimpanzees in a pristine habitat years before it was destined to be logged (Morgan *et al.*, 2006). Several orangutan studies were established in primary habitat that has since been logged, allowing retrospective analysis (e.g. Hardus *et al.*, 2012). However, habituation is not usually feasible or desirable in areas that are to be exploited on an industrial scale.

While our understanding of the general ecology of apes is good, being some of the best-studied taxa globally, the details of how resource extraction impacts ape ecology are still poorly known. Based on current knowledge of the behavior and ecology of apes in undisturbed natural environments, we are able to predict that extractive industries cause behavioral changes with subsequent physiological changes, but the impacts of these changes are hard to quantify. This is due to the complex relationships between extractive industry activities, their impact on resident apes' resource base, and the adaptive flexibility of each ape taxon to that impact within a specific environment. Thus these issues will be industry, site, and species specific, making it difficult to draw general principles. However, it is generally accepted that reductions in resource abundance are likely to, at best, drive changes in behavior of resident apes as they adapt to the changed quality, quantity, and distribution of resources. At worst we could expect increased levels of stress, reduced energy budgets, immuno-

Photo: While our understanding of the general ecology of apes is good, being some of the best-studied taxa globally, the details of how resource extraction impacts ape ecology are still poorly known. © Takeshi Furuichi, Wamba Committee for Bonobo Research



suppression, and increases in disease and parasite loads, leading to increased mortality and lower fertility. These impacts, together or independently, if sustained, are likely to be detrimental to the long-term viability of ape populations. Our understanding of recovery post-extraction is also poor, but it is clear that recovery will be determined by the ecology of the resident ape taxon, as well as extraction history, and the restoration regime.

Getting a better understanding of the complex socioecological responses of apes to resource extraction will require focused research using emerging techniques. The practical challenges of assessing the physical condition of apes in their natural habitat are enormous and until recently many of the physiological changes we might expect, stress in particular, could only be measured using invasive techniques. However, during the past decade huge strides have been made in the development of non-invasive sampling techniques and state-of-the-art diagnostics. Hormones, ketones, antibodies, pathogens, and parasites can now be extracted from feces and urine (e.g. Leendertz *et al.*, 2004; Gillespie, 2006; Masi *et al.*, 2012), making research on stress, reproductive endocrinology, diet, and nutritional status of wild animals feasible (e.g. Bradley *et al.*, 2007; Deschner *et al.*, 2012; Muehlenbein *et al.*, 2012; Murray *et al.*, 2013). Nonetheless, it will take studies of several generations of apes to elucidate how the stress, ranging variations, and behavioral changes induced by extractive industries impact their health and ultimately determine the survival, fecundity, stability, and maintenance of their populations.

Conclusion

Beyond broad generalities, little precise information exists on the ecological needs of apes in relation to specific forest attributes, as little is known of the normal or stochastic variations in the distribution and abundance

of most ape species. Also, few detailed quantitative data are available on how direct impacts differ, other than scale, therefore no simple inferences can be drawn about the impacts of extractive industries on apes. Specific studies are needed to establish baselines against which to assess impacts. These will include, but not be limited to, surveying ape populations at regular intervals to detect changes in their abundance and distribution. Targeted and frequent monitoring should produce the data needed to support more effective decision-making and adaptive management in concessions and surrounding buffer zones.

Carrying out baseline studies of ape populations often requires considerable support from the extractive industry. This, in turn, requires the industry to be either willing or coerced to provide such support, particularly during the early stages of a project when financial resources are limited as company investment is tied into exploration activities to ensure there is a profitable resource for exploitation. Logging is different but again, company investment is frequently channeled into infrastructure for extracting logs rather than carrying out surveys or EIAs. Thus there is a clear and pressing need for (1) education of the extractive industries, so that they understand the importance of early stage studies, and (2) enforced regulatory regimes or incentives which actually encourage companies to implement the recommended studies and mitigation measures. Voluntary action is not sufficient, therefore laws or incentives intended to change company behavior are a key missing element. As with the indirect impacts of the extractive industry, key issues are weak governance, inconsistent government policies, insufficient resources, a lack of enforcement, and corruption. The allocation of permits for exploration and extraction must include legal requirements for the adoption of wildlife-friendly and social best practices before, during, and after exploration/extraction has

TABLE 3.2**Potential impacts of extractive industries on apes**

Industry: project phase	Expected responses	
	Chimpanzees and bonobos	Gorillas
POTENTIAL IMPACT: Large-scale loss of habitat (expected in cases of open cast mining and logging)		
LSM: I, O	High death rates, especially infants and weaker individuals, due to starvation or reduced food intake	High death rates, especially infants and weaker individuals, due to starvation or reduced food intake
ASM: E, I, O	Limited, restricted and reduced feeding opportunities	Limited, restricted and reduced feeding opportunities
O and G: I, O	Elimination of nesting sites	Reduction in number and quality of nesting sites (ground and trees)
SL:	Breakdown or total collapse of community structure	Females possibly integrated into other groups
	Destabilization of surrounding communities	Destabilization of groups with silverback males fighting for dominance as group is displaced
	Integration of females into other communities	Possible increase in disease as animals are weakened by hunger
	Death of males (especially the alpha male) due to intercommunity conflict (less likely with bonobos)	
	Increased conflict over reduced resources	
	Possible increase in disease as animals are weakened by hunger	
POTENTIAL IMPACT: Partial loss and fragmentation of habitat		
LSM: E, I, O, C	Limited, restricted and reduced feeding opportunities	Limited, restricted and reduced feeding opportunities
ASM: E, I, O, C	Degradation/reduction of home range	Degradation/reduction of home range
O and G: E, I, O, C	Breakdown and possible fragmentation of community	Breakdown or possible fragmentation of group
SL:	Elimination of nesting sites	Reduction in number and quality of nesting sites (ground and trees)
	Breakdown or total collapse of community structure	Females possibly integrated into other groups
	Destabilization of surrounding communities	Destabilization of groups with silverback males fighting for dominance as group is displaced
	Integration of females into other communities	Possible increase in disease as animals are weakened by hunger
	Death of males (especially the alpha male) due to intercommunity conflict (less likely with bonobos)	
	Increased conflict over reduced resources	
Possible increase in disease as animals are weakened by hunger		
POTENTIAL IMPACT: Habitat degradation/reduction (e.g. noise, reduced air or water quality, change in habitat composition)		
LSM: E, I, O, C	Disruption of home range delineation	Disruption of home range delineation
ASM: E, I, O, C	Possible reduction in food sources due to invasive species and loss of total habitat area	Reduction in food sources due to invasive species and loss of total habitat area
O and G: E, I, O, C		
SL:		

Industry: project phase	Expected responses	
	Gibbons	Orangutans
POTENTIAL IMPACT: Large-scale loss of habitat (expected in cases of open cast mining and logging)		
LSM: I, O	High death rates, especially infants, juveniles, and weaker individuals, due to starvation or reduced food intake	High death rates, especially infants and weaker individuals (particularly females because they are more philopatric), due to starvation or reduced food intake
ASM: E, I, O	Limited, restricted and reduced feeding opportunities	Reduced feeding opportunities (change in diet, likely less caloric intake)
O and G: I, O	Reduced population density	Reduction in number of nesting sites (trees)
SL:	Changes in ranging behavior	Males moving out of cleared areas
	Changes in activity budgets to an energy conservation strategy	Possible increase in disease as animals are weakened by hunger
	Increased conflict with neighboring groups if displaced during operations	Shifts in home range use
	Possible increase in disease as animals are weakened by hunger and increased stress	Increased conflict over reduced resources (predominantly between females)
		Reduction in female reproductive rates due to lower food availability
		Reduction in home range size
		Change in time budget (more travelling, less feeding, less resting)
	Reduction in social behavior due to fewer opportunities for large party sizes due to reduced food	
POTENTIAL IMPACT: Partial loss and fragmentation of habitat		
LSM: E, I, O, C	Limited, restricted and reduced feeding opportunities	Reduced feeding opportunities (change in diet, likely less caloric intake)
ASM: E, I, O, C	Reduced population density	Reduction in home range size
O and G: E, I, O, C	Degradation/reduction of home range	High death rates, especially infants and weaker individuals (particularly females because they are more philopatric), due to starvation or reduced food intake
SL:	Increased mortality from falls	Reduction in number of nesting sites (trees)
	Population isolation and loss of population viability in smaller fragments	Males moving out of cleared areas
	Reduced dispersal options	Possible increase in disease as animals are weakened by hunger
	Possible increase in disease as animals are weakened by hunger	Shifts in home range use
		Increased conflict over reduced resources (predominantly between females)
		Reduction in female reproductive rates due to lower food availability
	Change in time budget (more travelling, less feeding, less resting)	

Industry: project phase	Expected responses	
	Gibbons	Orangutans
POTENTIAL IMPACT: Habitat degradation/reduction (e.g. noise, reduced air or water quality, change in habitat composition)		
LSM: E, I, O, C	Disruption of home range delineation	Reduced feeding opportunities (change in diet, likely less caloric intake)
ASM: E, I, O, C	Possible reduction in food sources due to invasive species and loss of total habitat area	Reduction in home range size
O and G: E, I, O, C		High death rates, especially infants and weaker individuals (particularly females because they are more philopatric), due to starvation or reduced food intake
SL:		Reduction in number of nesting sites (trees)
		Males moving out of cleared areas
		Possible increase in disease as animals are weakened by hunger
		Shifts in home range use
		Increased conflict over reduced resources (predominantly between females)
		Reduction in female reproductive rates due to lower food availability
		Change in time budget (more travelling, less feeding, less resting)

Notes:

Extractive industry: LSM = large-scale mining, ASM = artisanal and small-scale mining, O and G = oil and gas development, SL = selective logging

Project phase: E = exploration, I = implementation, O = operation, C = closeout

occurred (see Chapter 7 for more information and examples).

Nearly a decade of continuous research in the Goualougo Triangle has demonstrated that gorillas and chimpanzees can co-exist with RIL (D. Morgan, C. Sanz, S. Strindberg, J. Onononga, C. Eyana-Ayina, and E. Lonsdorf, personal communication, 2013). Likewise, one detailed longitudinal study on gibbons suggests that gibbon populations can persist and rebound in selectively logged areas under particular circumstances (Johns, 1986a; Johns and Skorupa, 1987; Grieser Johns and Grieser Johns, 1995); however, the conditions required for population persistence remain unknown. A few studies have noted that Sumatran orangutans are less tolerant of logging, possibly due to their more specialized dietary requirements (Husson *et al.*,

2009; Hardus *et al.*, 2012). Bornean orangutans appear to survive outside protected areas such as in the FSC certified concession, Dermakot in Sabah, Malaysia, at the present time or at least in the short term (see also Marshall *et al.*, 2006; Ancrenaz *et al.*, 2010).

However, it is too soon to comment on long-term survival in timber estates with respect to this long-lived, slow-reproducing species. Of all the forms of mechanized logging, certified timber operations seem to be the most compatible with the persistence of apes for a variety of reasons. Ensuring the long-term viability of apes requires greater emphasis to be placed on maintaining the quality and quantity of their food and nest resources in relation to forestry treatments.

Overall, the available evidence suggests that conventional logging negatively impacts biodiversity, but that sustainably managed forests can maintain viable populations of apes and therefore contribute to their conservation. However, it is important to stress that concessions are not a substitute for unlogged primary forests and the protected area network (Clark *et al.*, 2009; Gibson *et al.*, 2011; Woodcock *et al.*, 2011). Consequently, proximity of unlogged suitable habitat plays a vital role in both the short- and long-term survival prospects of apes in modified habitats. Such areas provide “refuge” and effectively buffer some animals from negative impacts, although details such as optimal distance to refuge areas or characteristics signifying the quality of these habitats are unknown.

Despite the variability observed, severity of the impacts of logging on apes seems to be a factor of (1) type of logging practice, (2) availability of adequate undisturbed and suitable habitat adjacent to logging sites, (3) intensity of logging, and (4) control of associated activities, such as hunting and clearing of land for agriculture. Ape populations appear to be able to recover if the right mitigating factors can be assured. Additionally, shifts in resource use and behavior observed across a continuum of human influence highlight the flexibility of these apes in adapting to environmental changes and opportunities (Hockings, Anderson, and Matsuzawa, 2006, 2012; Meijaard *et al.*, 2010; D. Morgan, C. Sanz, S. Strindberg, J. Onononga, C. Eyana-Ayina, and E. Lonsdorf, personal communication, 2013). Such observations are encouraging.

In the long term, the impacts of extractive industries on apes will depend on how well a company: (1) understands the ecological and behavioral requirements of resident apes, especially for shelter, food, both social structure and social dynamics, and space; (2) recognizes the potential threats to resident apes from logging or the operational practices during all phases of a mining/oil and

gas project; and (3) identifies and manages potential biodiversity risks and opportunities during the relevant phases of the project. These are described in more detail in both Chapters 4 and 5.

It is incredibly important for industries to recognize the immediate and enduring impacts that individual projects can have on ape populations and associated biodiversity. Avoidance and mitigation of negative impacts is always more effective and less costly than repair or offsets. RIL and certification of logging operations are examples of effective approaches that may reduce the negative impacts on apes. The actions already being taken by some companies to apply technologies to anticipate and reduce potential impacts and to carry out mitigation measures that will avoid and minimize the negative impacts must be applauded and held up to serve as essential lessons to guide ape conservation strategies.

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