Synanthropic survival: low-impact agriculture and White-shouldered Ibis conservation ecology

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Abstract

The conservation value of traditional agriculture is well recognised in Europe, where retention and restoration of farming practices that support open-habitat species is a standard management technique. Elsewhere, however, this value is often overlooked while conservation attention is directed at natural habitats and forest biota. This thesis assesses the importance of traditional farming for developing-world biodiversity, using the White-shouldered Ibis *Pseudibis davisoni* in Cambodia to investigate practices underpinning synanthropic relationships, links between farming-dependent species and local livelihoods, and potential conservation strategies. Ibis status and ecology was investigated by censuses, foraging observations, prey sampling, experimental exclusion of grazing and burning at foraging habitats, and experimental protection of nests. Livelihoods were assessed by social research methods including household income surveys. A literature review found a subset of threatened bird taxa now dependent on traditional farming following the loss of natural processes. Agricultural change, driven by external agribusiness and intrinsic livelihood modernisation, endangers these species, including the ibis. Ibis foraging ecology is closely associated with local livelihood practices, with favoured dry forest habitats created or maintained by domestic livestock grazing, anthropogenic fire and rice cultivation. Not all local practices are beneficial, however: ibis nests are exploited for food by local people, and nest guardians do not improve nest success (although this requires further testing). White-shouldered Ibis's breeding season contrasts with that of the sympatric Giant Ibis *Thaumatibis gigantea*, most likely explained by the former's dry-season-adapted foraging strategy. Household incomes and livestock capital assets demonstrated that local people share a dependence on the livelihood practices and dry forest landscape supporting the ibis. Nevertheless, local livelihood change (such as mechanisation) may uncouple this linkage, making a potential win-win conservation strategy unviable. Conservation must develop measures to maintain valuable farming practices before they, and the species dependent on them, are lost through agricultural transition.

Contents

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Chapter 1

Introduction

Rice cultivation remains traditional in northern Cambodia, with harvesting done by hand.

1.1. Challenges for agriculture, challenges for conservation

> *"to address the poverty of a billion people not getting enough food, [and] with another billion [in population growth] in 13 years' time, you've got to massively increase agriculture".*
>
> Sir John Beddington, Chief Scientific Advisor, March 2012.

The recent report of the Commission on Sustainable Agriculture and Climate Change (Beddington et al. 2012) has brought renewed attention to agriculture and its capacity to meet higher demands for food, vegetable oil and energy crops in the next four decades. Increasing consumption, driven by population growth and escalating wealth (Godfray et al. 2010), will necessitate a “massive increase” in agricultural output. Achieving this sustainably is a major concern (McLaughlin 2011; Tilman et al. 2011; Tilman et al. 2002). Reducing agriculture’s environmental costs has already proved difficult: after several decades spent documenting biodiversity declines (Carson 1962; Donald 2004; Tucker & Heath 1994) and attempting to mitigate its impacts (Balmford et al. 2005a; Kleijn & Sutherland 2003), the sector remains the most damaging to nature (Balmford et al. 2012; MEA 2005). Reconciling biodiversity protection with accelerating crop demand is now a key challenge for conservationists (Balmford et al. 2005b), and the likely scale and impact of agricultural growth calls for better integration of conservation and farming (Adams 2012; Norris 2008).

1.2. Paradigms of agriculture in conservation

Agriculture’s place in conservation differs between the developed world, particularly Europe, and the developing world. As most of Europe’s ecosystems are already radically transformed by agriculture (Donald et al. 2002), protecting the nature value inherent to farmed and semi-natural landscapes has become a paradigm of European conservation (Sutherland 2004). Declines in many farmland taxa (Donald et al. 2001; Pywell et al. 2006; van Swaay et al. 2006) have provoked considerable attempts to integrate conservation into farming, most notably through European Union agri-environmental schemes (Kleijn & Sutherland 2003). In parallel, conservation of valuable human-modified, semi-natural habitats (such as heathland, grassland and fens) frequently adopts low-intensity farming techniques, many of which have benefited or accommodated biodiversity for centuries or even millennia (Bignal & McCracken 2000; Kleijn et al. 2006; Sutherland & Hill 1995).

Elsewhere conservation takes a different viewpoint, particularly in tropical countries of the developing world, where conserving “wild nature”, in more intact ecosystems, takes priority. Agriculture and conservation are often considered incompatible in this context (Tscharntke et al. 2005) as the habitat devastation visible along many agricultural frontiers generates widespread concern for natural integrity. This shapes conservation foci in the developing world, with greater attention given to forests rather than other, more open biomes (Bond & Parr 2010). However, agriculture’s value does receive attention in the paradigm of countryside biogeography (Daily et al. 2001), which focuses on improving the agricultural matrix to support (usually forest) species in remnant natural habitat patches (Perfecto & Vandermeer 2010; Vandermeer & Perfecto 2007). In contrast to Europe, where conservation promotes agricultural practices benefiting open-habitat species [1], efforts in the developing world generally aim to minimise the impacts of agriculture threatening closed-habitat species, either by reducing forest conversion and/or degradation, or improving functional connectivity in fragmented landscapes.

1.3. A new research agenda

New research considers the strategies for enabling increased agricultural production alongside biodiversity conservation. Land-sparing and land-sharing are two contrasting options proposed (Fischer et al. 2008; Green et al. 2005). Land-sparing would increase yields on existing farmland, reducing the need to convert new land for agriculture and thereby sparing land for conservation. Land-sharing advocates wildlife-friendly practices to maintain biodiversity within farmland, but likely costs to yield will require that more land becomes cultivated (Phalan et al. 2011a) if demand for agricultural output cannot otherwise be alleviated. The relative benefits of these strategies remain contested (Adams 2012; Fischer et al. 2011; Phalan et al. 2011b), but the debate has promoted a new research agenda into the compatibility of agriculture and conservation. Conservation scientists now seek holistic, interdisciplinary approaches to understand: biodiversity retention in farmland; valuable farming methods; the ecological, social and political conditions that suit alternative conservation strategies; and mechanisms to integrate conservation into agricultural policy (Balmford et al. 2012). Amongst the knowledge gaps is a need to understand which and what types of species benefit from agriculture and the mechanisms that underlie these patterns (Adams 2012; Norris 2008).

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[1] Open-habitat species are those that once occurred naturally in non-forested biomes such as grasslands, savannas, and steppe.

1.4. Synanthropy in agricultural landscapes

Benefits of agriculture are particularly apparent in semi-natural [1] or extensively farmed [2] landscapes of the developed world, where many species have become closely associated with human activity (synanthropy). Over several centuries and even millennia, many species followed the spread of open habitats as agricultural land use expanded (Donald et al. 2002), resulting, in combination with the loss of natural processes, in strong relationships with agriculture. Farmland resources and ecological functions have become vital to a range of open-habitat taxa (Attwood et al. 2009; Michael et al. 2011; van Swaay et al. 2006) and land-management techniques such as livestock grazing, burning and crop rotation, combined with minimal chemical and mechanical input, create high conservation value (Bignal & McCracken 1996). Many birds, for example, have come to rely on the high invertebrate densities, weed seeds, crop residues and spilt grain, animal carcasses and spatial and temporal habitat heterogeneity provided by low-impact [3], and often traditional, forms of farming (Fuller et al. 2004; van der Weijden et al. 2010). Population declines with twentieth-century agricultural modernisation are indicative of many species' present-day dependencies on low-impact farming.

Agriculture-dependent species demonstrate a unique nature value inherent to low-impact farming systems, but much remains to be learnt about this subset of biodiversity. Examples are most apparent from the developed world, particularly Europe, but with prevalent research paradigms directing little attention to agricultural landscapes elsewhere, these phenomena may reflect more than one continent's idiosyncratic ecology. Isolated cases

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[1] Semi-natural habitats/landscapes are defined for this thesis as those that contain a near-natural selection of species but are modified and, at least in part, sustained by human activity, so that if management is removed, the habitat and its species assemblage would likely change e.g. through succession.

[2] Extensive farming are modes of production that require little or no labour, chemical or capital inputs relative to the land area in use. Extensive pastoral farming involves no chemical treatment of pastureland and has low stocking densities, often over large land areas.

[3] Low-impact farming/agriculture is defined, for the purposes of this thesis, as modes of production that have little to moderate ecological impact, therefore minimising the loss of species that occur naturally, or have become long-established components of the farmed landscape. These modes typically make no or little use of chemical treatments and advanced farm machinery, instead adopting cropping and livestocking techniques that, as a by product of farming, maintain or enhance resources for wildlife. These farming systems are often a precursor to, and contrast strongly with, the highly mechanised, high-input modes of agriculture now prevalent in much of the developed world (particularly Europe and North America).

from further afield – such as the Sociable Lapwing *Vanellus gregarius* in Central Asia (Kamp et al. 2009) – and the long history of agriculture in many parts of the developing world (Mazoyer & Roudart 2006) suggest synanthropy may not be as uncommon outside of Europe as has been widely perceived. If this important subset of biodiversity is widespread, whether it can persist in farmed landscapes undergoing technological change, or be conserved alongside the development needs of local people, are important questions (Adams 2012). With increased global production driving agricultural modernisation (Horlings & Marsden 2011), there is an urgent need to identify the conservation value in traditional, low-impact farming landscapes, and understand the practices that sustain agriculture-dependent biodiversity, before they are irreversibly changed.

1.5. White-shouldered Ibis: a case study

This thesis focuses on the example of White-shouldered Ibis *Pseudibis davisoni* in a traditional, mixed farming system of Cambodia. As data for a wide range of synanthropic species is not yet available, this case study is useful to illustrate conservation issues surrounding an agriculture-dependent species in the developing world. This Critically Endangered ibis was selected for this purpose as it: occurs in a wildlife-rich, yet poorly studied, farming system comprising both low-intensity arable and extensive pastoral agriculture; is confined to a region likely to undergo imminent, substantial agricultural change (Yu & Diao 2011); and, given its severe endangerment, requires urgent research to understand its links with farming.

Prior to this study, crude estimates put the global White-shouldered Ibis population at only 50-250 mature individuals, following a dramatic decline in the twentieth-century (BirdLife International 2008). Reasons for this population crash are uncertain, but hunting and habitat loss appear likely factors (BirdLife International 2001), leaving it the most threatened waterbird in South-East Asia (Tordoff et al. 2005). Although functionally extinct from Thailand, Myanmar, southern China and Vietnam, and very scarce in southern Laos and Indonesian Borneo (BirdLife International 2001; Meijaard et al. 2005), the rediscovery of subpopulations in dry forests of north and east Cambodia confirmed a final stronghold (BirdLife International 2002; WCS 2004).

Historic records of White-shouldered Ibis indicate use of wetlands, river channels and cultivated lands (BirdLife International 2001). Anecdotal evidence found the species in closer proximity to people than other South-East Asian large waterbirds (J.C. Eames and T.

Evans pers. comm. 2008), provoking suggestions that species is synanthropic and reliant on foraging habitat grazed by livestock (Buckingham & Prach 2006). Timmins (2008) postulated that declines in wild ungulates may have altered wetland grazing ecology and impacted ibis populations, and a study of dry season foraging ecology confirmed the ibis's selection of pools with short vegetation, plus the use of forest understorey where bare ground was available (Wright 2008; Wright et al. 2010). Available data suggested that the ibis is a dry-season breeder vulnerable to nest robbery and disturbance (Clements et al. in press-a), but scientific evidence for most of its foraging and breeding requirements was still lacking at this study's inception. A workshop was held with governmental and non-governmental conservation organisations in Phnom Penh, February 2009, to identify and prioritise the knowledge gaps in White-shouldered Ibis ecology.

1.6. Thesis background

1.6.1. Thesis objectives

This thesis seeks evidence for numerous species depending on developing-world agriculture, drawing attention to the value of tropical and developing-world farming landscapes neglected in conservation science. By revealing this distinctive subset of biodiversity, the thesis aims to inform our knowledge of biodiversity retention in agricultural landscapes, bring new considerations for the land-sparing versus land-sharing debate, and deliver wider relevance for the semi-natural habitats paradigm in European conservation. The thesis considers the case of White-shouldered Ibis to exemplify some of the ecological mechanisms that underpin synanthropic relationships with agriculture. The importance of a traditional farming system to local livelihoods is assessed in an attempt to find synergies between ibis conservation and human well-being; likely impacts of socio-economic change are also evaluated. In addition, the thesis presents data on White-shouldered Ibis population status and foraging ecology – informative to conservation and the study of this species's synanthropy – and tests the effectiveness of nest-guarding, a popular but poorly studied intervention that engages local communities in conservation.

1.6.2. Research approach

Relationships between humans and biodiversity are particularly evident in agriculture, where people and wildlife rely on the same land area, ecosystem services, and often each other. Synanthropic species provide a prime example of the need to integrate conservation and development perspectives, as changes to farming livelihoods will affect conservation intervention and vice versa. This study therefore adopts the interdisciplinary approach now widely called for in conservation science (Balmford et al. 2012; Campbell 2005; Norris 2008), conducting ecological research alongside livelihood assessment and rural appraisal methods frequently applied in social sciences.

1.6.3. Study areas

1.6.3.1. Cambodian dry forests

Central Indochina was once dominated by deciduous dipterocarp forests (DDF), but following deforestation the largest areas remain in north and east Cambodia (CEPF 2007). Dry forest landscapes comprise a DDF matrix surrounding a mosaic of grasslands, mixed deciduous and semi-evergreen forests, river channels, and active and abandoned rice paddies. DDF is typically open in structure, lacking a shrub or middle-storey canopy and resembling a savannah. The climate is strongly monsoonal with average monthly rainfall as little as 0.9 mm in the dry season (November–April) and up to 333 mm in the wet season (May–October; Thuon & Chambers 2006). Waterholes, known locally as *trapaengs* and of 0.001–3.4 ha (Fig. 1.1.; Wright et al. 2010), occur extraordinarily frequently in the landscape. *Trapaeng* and river channel water levels vary seasonally (Thuon & Chambers 2006; Wright et al. 2010), with water drawdown exposing pool and river-bed substrates in the dry season.

Small villages occur sporadically in the landscape and local people practice low-intensity, wet-season rice cultivation, supplemented by extensive livestock rearing (namely cattle and domestic water buffalo) and harvesting of dry forest resources (Clements et al. in press-b; McKenney & Prom 2002). Poverty is widespread with household consumption in Cambodian forests estimated at only $329 ± 16 per annum (mean ± SD; World Bank 2009). Livestock are released into the forest to feed for most of the year; both people and livestock permeate the majority of the landscape but their abundance declines with distance from villages (Wright et al. 2010). The forest understorey is burnt annually to encourage new graze for their livestock (Stott 1986). Livestock grazing and anthropogenic fire are both likely to affect the suitability of foraging habitat for ibis.

Figure 1.1. A *trapaeng* in the mid-dry season, visited by domestic cattle and water buffalo and showing exposure of heavily grazed and trampled substrate.

Dry forests once supported large populations of mega-fauna, including four wild bovids and Asian Elephant *Elephas maximus* (Tordoff et al. 2005; Wharton 1968). Although these populations have much declined (Loucks et al. 2009), these landscapes still support at least 30 threatened mammals and 19 threatened birds, including the White-shouldered Ibis and the similarly-threatened Giant Ibis *Thaumatibis gigantea* (WCS 2009). While livelihood activities do have an impact (e.g. logging, clearance for agriculture and hunting), larger-scale habitat conversion for economic land concessions (often plantation agriculture), infrastructure and settlement are the most serious threats to the dry forest ecosystem, and to White-shouldered Ibis (BirdLife International 2012; CEPF 2007).

1.6.3.2. Western Siem Pang Important Bird Area

The principal study site was Western Siem Pang Important Bird Area (IBA; Seng et al. 2003) in Stung Treng province, northern Cambodia (14°07'N 106°14'E; Fig. 1.2.). This 138,000 ha area holds the largest known White-shouldered Ibis population – believed, before this study, to number at least 140 birds (D. Buckingham unpubl. data) – and contains a typical selection of dry forest habitats. Legal protection has not yet been secured

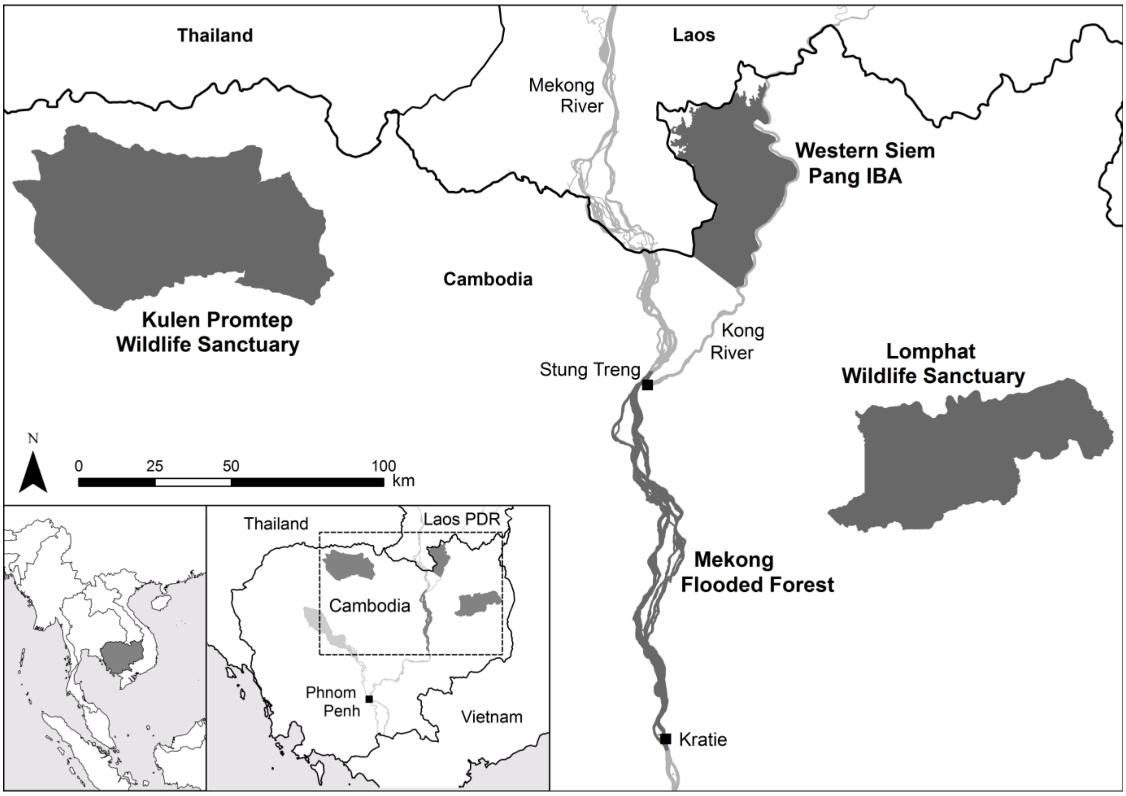

Figure 1.2. Map of the four main study sites in north and east Cambodia; insets show the regional context with Cambodia highlighted in grey in the left-hand inset. Black squares indicate provincial centres of interest.

(Timmins 2012), but BirdLife International have undertaken small-scale conservation activities since 2003, employing four staff to undertake basic biodiversity monitoring and local awareness campaigns (BirdLife International 2009). Sporadic law enforcement, coordinated with the Forestry Administration, has focused on illegal logging. Approximately 11,000 people live in 16 small settlements (Ministry of Planning 2007); the nearest provincial capital (and large market) is *c*. 75 km from the IBA's centre and, until 2010, was reachable only by seasonally-passable forest tracks, or by boat along the Kong River (Fig. 1.2.). The stretch of this river immediately adjacent to Western Siem Pang (forming part of the Sekong River IBA; Seng et al. 2003) was studied for chapter 5.

1.6.3.3. Other study sites

Research was undertaken at three other sites (Fig. 1.2.) to estimate White-shouldered Ibis population size and nesting success only. Two of these were protected, dry forest areas:

Kulen Promtep Wildlife Sanctuary, Preah Vihear province (13°58'N 104°53'E), where a variety of conservation interventions have been undertaken since 2002 (Clements et al. 2010), and Lomphat Wildlife Sanctuary, Ratanakiri province (13°20'N 106°56'E), where law enforcement began in 2003, and species conservation activities in 2010 (BirdLife International 2010). A further site, the Mekong Flooded Forest in Kratie and Stung Treng provinces (13°02'N 106°01'E), is an unprotected area comprising the braided channel of the Mekong River, surrounded by DDF, mixed deciduous and semi-evergreen forests; conservation activities have taken place since 2010. Further details of these sites are provided in the relevant chapters.

1.6.4. Thesis structure

The thesis begins with a discussion of the biodiversity value in developing-world agriculture (chapter 2), presenting evidence for an assemblage of species depending on agriculture. Subsequent chapters study White-shouldered Ibis, with chapters 3–5 setting the ecological context. Chapter 3 presents results of coordinated roost counts, estimating White-shouldered Ibis population size and assessing the proportion protected within formally designed sites. Chapter 4 examines White-shouldered Ibis foraging ecology at *trapaengs*, considering how the species overcomes water scarcity to breed in the dry season. Chapter 5 compares the foraging strategies of sympatric White-shouldered Ibis and Giant Ibis and discusses how their use of the mixed farming system may differ. Chapter 6 experimentally tests the impact of livestock grazing and forest understorey fires on White-shouldered Ibis foraging habitats, establishing the link between traditional farming practices and ibis ecology. Chapter 7 examines local livelihoods, ascertaining local people's reliance on dry forest resources and farming practices valuable to ibis, and discussing the potential for a win-win conservation approach. Chapter 8 considers the effectiveness of locally-employed nest guards and the contexts in which this conservation intervention is most useful. Chapter 9 concludes with a summary and discussion of the thesis findings.

The seven results chapters (chapters 2–8) are written in the form of scientific peer-reviewed papers. At the time of submission, three chapters were published: chapter 2 (Wright et al. 2012c), with an accompanying correspondence piece (Wright et al. 2012d) given in the chapter's Appendix D; chapter 3 (Wright et al. 2012b); and chapter 5 (Wright et al. 2012a).

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Chapter 2

Agriculture – a key element for conservation in the developing world

Sociable Lapwing *Vanellus gregarius*, a threatened steppic species reliant on farming. Photo courtesy of Manjeet & Yograj Jadeja.

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2.1. Abstract

Conserving biodiversity through supporting or mimicking traditional management of anthropogenic habitats is a paradigm in the developed world, particularly Europe. It is rarely applied in developing countries where forest biotas are more common foci. We quantified the numbers of globally threatened bird species using anthropogenic habitats and examined scientific literature to identify those that are dependent on low-impact agriculture in the developing world. Such dependency is distinct from species using farmland to supplement or move between their remnant natural habitats. We show that low-impact agriculture is important to a number of threatened open-habitat species in a variety of farming systems. However, these systems are expected to undergo widespread transformation due to economic change. Conservation must identify valuable farmed landscapes and seek new mechanisms to maintain or mimic important land-management techniques in developing countries. A suite of policy instruments should be considered, to provide incentives or development benefits that encourage farmers to manage landscapes for wildlife. The land sparing approach to balancing biodiversity conservation and agricultural production will be detrimental to those open-habitat bird species dependent on agriculture; a mix of agricultural land-use types may offer the best compromise.

2.2. Introduction

Anthropogenic landscapes are receiving increasing attention in developing world conservation (Daily 2001; Edwards et al. 2011; Gardner et al. 2009; Urquiza-Haas et al. 2007). However, outside of developed countries the conservation value of the agricultural matrix is usually interpreted in terms of its permeability to forest species that retain access to fragments of natural habitat (Daily et al. 2001; Perfecto & Vandermeer 2010; Ranganathan et al. 2008). Here we present evidence that anthropogenic landscapes are of primary importance to a distinct set of mainly open-habitat species. This situation most commonly arises in ancient farmed landscapes in parts of Europe, Africa and Asia, but also in recently transformed landscapes where novel human land use has substituted for natural ecosystem processes. Forest species, the focus of countryside biogeography, use farmlands as a means of dispersal, a buffer to populations in core natural ecosystems or as a supplementary resource. For these, farmland is still only second best compared to intact forest. In contrast, many open-habitat species have come to fully depend on anthropogenic or semi-natural landscapes where their natural habitat has been entirely lost. Examples of this dependency can now be found in the developing world, where conservation approaches emphasising semi-natural landscapes will have great relevance.

2.3. A developed world conservation paradigm

Anthropogenic landscapes sustain much biodiversity in the developed world, following the loss and conversion of natural ecosystems over recent millennia. Although progressive landscape transformation extirpated numerous species (often filtering top predators, large herbivores, old-growth dependent and some open-habitat species), remaining taxa were able to exploit these landscapes and the low-impact practices that maintained them, resulting in dependency in the absence of their natural habitat. Open-habitat species, those once occurring naturally in non-forested habitats such as grasslands, savannas and steppe, have developed particularly strong dependencies on anthropogenic and semi-natural habitats.

Twentieth-century mechanization and market transformation brought further ecosystem change (Donald et al. 2001). Intensified land use resulted in temporal and spatial homogenization of habitats (Benton et al. 2003), while abandonment of marginal lands caused ecological succession and further reductions in habitat complexity (Sirami et al. 2008). As a result, reintroducing or mimicking low-impact practices to sustain the conservation value of semi-natural habitats became a dominant paradigm in European conservation (Bignal & McCracken 2000; Sutherland & Hill 1995). Traditional management of forest, fen, anthropogenic grasslands, shrublands and pseudo-steppe habitats has been widely applied and incorporated into legislation, such as the European Habitats Directive.

Agriculture in Europe became a particular focus of the semi-natural habitats paradigm. Heterogeneous agricultural mosaics offer benefits to numerous complementing species (Fuller et al. 2004), while other taxa require extensively farmed landscapes of less structural complexity. Legislation such as the European Common Agricultural Policy has incentivised wildlife-friendly, lower-impact farming to counter the twin threats of agricultural intensification and abandonment. However, such agri-environmental schemes sometimes achieve mixed or meagre success due to broad and shallow approaches that minimise transaction costs at the risk of ignoring important ecological detail (Batáry et al. 2011; Kleijn et al. 2006).

2.4. Conservation and agriculture in the developing world

By contrast, the semi-natural habitat approach has rarely been applied in the developing world. While this may be partly due to challenging social and political conditions that limit policy transfer, dominant schools of thought in developing world conservation also contribute. Priority is given to closed-habitat species and their frontier forest ecosystems (Bond & Parr 2010), where agricultural conversion causes considerable primary habitat and species loss (Sodhi et al. 2010). Policy is dominated by efforts to stem the impacts of exploitation or land-use change in natural habitats. Agricultural landscapes, when considered, are typically assessed for their suitability in maintaining or assisting the survival of forest species, such as studies of wildlife-friendly coffee plantations (Mas & Dietsch 2004). Although in some cases agriculture is treated as an intimate component of biodiversity conservation (Perfecto & Vandermeer 2010), the focus remains on sustaining populations of declining natural habitats. Conservation approaches directed primarily at frontier ecosystems or by countryside biogeography may overlook the importance of agricultural landscapes for open-habitat species.

Global food demand is increasing due to growing human population but also greater affluence and changing consumption. Though famine and food security may best be addressed by resolving food entitlement inequalities (Sen 1981), global demand may nonetheless double by 2050, outstripping human population increase (Loh 2002). How this can be met without widespread species extinctions is of great concern, with agricultural land-use considered one of the greatest threats to global biodiversity (MEA 2005; Sala et al. 2000). The majority of human population and economic growth is occurring in developing countries where pressures for natural habitat conversion and agricultural intensification are greatest and expected to escalate (Cincotta et al. 2000; Tilman et al. 2001).

Species already dependent, or increasingly reliant, on farmland due to loss of natural habitats are at particular risk from agricultural change. It is important to identify and protect those semi-natural habitats and agricultural landscapes of high conservation value in developing countries. Here we present evidence that agricultural landscapes support not just a filtered subset of the biota remaining in extant natural habitats, but rather a unique and dependent biodiversity. This justifies the wider application of European conservation approaches to the developing world and influences how habitats are prioritised for conservation in these countries.

2.5. Threatened species and low-impact agriculture in the developing world

Globally threatened birds were systematically examined to quantify their associations with agricultural habitats, followed by an assessment of candidate species and their potential dependency on farming. The analysis was restricted to birds as there is little comparable autoecological data for other taxa. Nonetheless, the multitude of evidence from developed countries suggests that dependencies of non-avian wildlife on agriculture will also occur more widely. Numerous butterflies (van Swaay et al. 2006), arthropods (Di Giulio et al. 2001), reptiles (Michael et al. 2011), amphibians (Hartel et al. 2010), bats (Boughey et al. 2011) and even sessile organisms such as vascular plants (Haines-Young et al. 2000), rely on or benefit from management of anthropogenic habitats in the developed world.

Focusing on species of high conservation priority revealed the importance of agriculture to conservation globally. Habitat associations were collated and quantified across six regions: Europe, North America, Australasia, Asia, Africa and South America. We searched the *IUCN Red List for Birds* database (BirdLife International 2011) using terms consistently used for status (Critically Endangered, Endangered, Vulnerable or Near Threatened) and habitat (forest, grasslands, savannas or terrestrial artificial landscapes - which we interpret as mainly comprising agriculture); these are elaborated in Appendix A (section 2.9.1).

The potential agricultural dependency of candidate bird species was initially assessed using species accounts of the Red List database (BirdLife International 2011), identifying birds that make use of food resources or habitat conditions (foraging or breeding) maintained by farming practices. Where these suggested possible dependency (replacing or substituting, rather than complementing natural habitats) we sought scientific evidence from primary literature. Species were considered largely or entirely dependent on agriculture where approximately > 75% of the population was reliant on an agricultural habitat or practice at one or more stages of its life history. Population data were obtained from species accounts or primary literature, but when unavailable the proportion of the species range with dependency was inferred qualitatively from distribution maps. Our assessment of agricultural dependencies will be incomplete, particularly for grassland or savanna species, where species accounts and past autoecological studies have often failed to recognise the dynamic nature of these systems and the crucial role of human land use. The true importance of agriculture to dependent species is therefore underestimated.

Although 77% of all threatened or Near Threatened bird species in developing countries use forest habitats, 28% use terrestrial artificial landscapes (22% in addition to forests and 6%

in artificial landscapes but not forests). Thirty-three percent of threatened species in Asia use artificial habitats (Fig. 2.1.), matched by 33% of African and 20% of South American species, demonstrating that such associations are widespread in the developing world. Furthermore, 25% of all globally threatened or Near Threatened developing-world birds occur in grassland or savanna habitats, many of which are modified or maintained by human land-use. Grassland is especially valuable in Africa, where it is used by 95 of the 144 globally threatened birds (Beresford et al. 2010).

Beyond the use of agricultural and potentially modified habitats presented in Fig. 2.1., we identified nearly thirty threatened bird species for which there is strong evidence of dependence on low-impact agriculture in the developing world (Table 2.1.). The number of examples suggests this is not a trivial pattern and many more cases would be found if appropriate data were available. We found dependence on anthropogenic landscapes and habitats across a wide range of open-habitat species and taxonomic groups, from grassland specialists such as larks and bustards, to birds of prey and waterbirds. These occurred at both breeding and non-breeding life stages and across all six geographic regions. As in Europe, open-habitat species worldwide benefit from a variety of resources and management techniques across a range of farming systems.

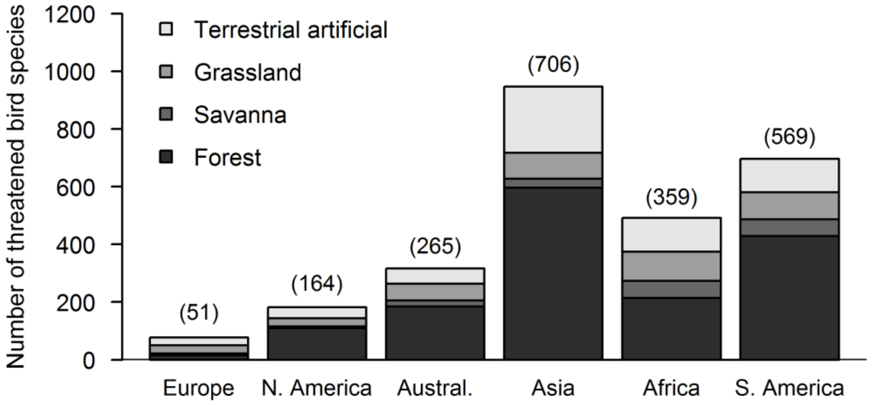

Figure 2.1. The numbers of globally threatened or Near Threatened species using forest, savanna, grassland or other artificial habitats (primarily agricultural but also including urban, rural gardens and heavily degraded forest) by region. As individual species may use more than one habitat and more than one region, the number of species represented in each region is shown in parentheses. Australasia (Austral.) includes Australia, New Zealand, New Guinea and Pacific Islands.

Farming system	Species		Resource [a]	Status [b]	Region [c]
Extensive pastoral	Jerdon's Courser	*Rhinoptilus bitorquatus*	FH	CR	South Asia
	Liben Lark	*Heteromirafra sidamoensis*	NH, FH	CR	East Africa
	Rudd's Lark	*Heteromirafra ruddi*	NH, FH	VU	Southern Africa
	Dupont's Lark	*Chersophilus duponti*	NH, FH	NT	North Africa
	Sharpe's Longclaw	*Macronyx sharpei*	NH, FH	EN	East Africa
	Sierra Madre Sparrow	*Xenospiza baileyi*	NH, FH	EN	Central America
Pastoral	Indian Vulture	*Gyps indicus*	Ca	CR	South Asia
	Slender-billed Vulture	*Gyps tenuirostris*	Ca	CR	South & South-East Asia
	White-rumped Vulture	*Gyps bengalensis*	Ca	CR	South & South-East Asia
	Red-headed Vulture	*Sarcogyps calvus*	Ca	CR	South & South-East Asia
	St Helena Plover	*Charadrius sanctaehelenae*	NH, FH	CR	South Atlantic islands
	Sociable Lapwing	*Vanellus gregarius*	NH, FH	CR	Central Asia
	Buff-breasted Sandpiper	*Tryngites subruficollis*	FH $_{NB}$	NT	South America
	Botha's Lark	*Spizocorys fringillaris*	NH, FH	EN	Southern Africa
	Pale-headed Brush-finch	*Atlapetes pallidiceps*	FH	EN	South America
Arable and rice	Asian Crested Ibis	*Nipponia nippon*	FH	EN	Central Asia
	Black-necked Crane	*Grus nigricollis*	FH, Gr $_{NB}$	VU	Central Asia
	Hooded Crane	*Grus monacha*	FH, Gr $_{NB}$	VU	East Asia [d]
	Yellow-breasted Bunting	*Emberiza aureola*	FH, Gr $_{NB}$	VU	Asia
Mixed pastoral and arable	Northern Bald Ibis	*Geronticus eremita*	FH	CR	North Africa, Middle East
	Southern Bald Ibis	*Geronticus calvus*	FH	VU	Southern Africa
	White-shouldered Ibis	*Pseudibis davisoni*	FH	CR	South-East Asia
	Blue Crane	*Grus paradisea*	NH, FH, Gr	VU	Southern Africa
	Grey Crowned-crane	*Balearica regulorum*	FH, Gr	VU	South & East Africa
	Bengal Florican	*Houbaropsis bengalensis*	NH, FH	CR	South & South-East Asia

Table 2.1. Continued pg. 29.

Blue Bustard	*Eupodotis caerulescens*	NH, FH	NT	Southern Africa
Great Bustard	*Otis tarda*	NH, FH	VU	Middle East, Central Asia [d]
Great Indian Bustard	*Ardeotis nigriceps*	NH, FH	CR	South Asia
Saffron-cowled Blackbird	*Xanthopsar flavus*	NH, FH	VU	South America

Table 2.1. Threatened and Near Threatened open-habitat species dependent on low-impact agriculture in the developing world, by farming system and in taxonomic order [1]. Sources are given in Appendix B (section 2.9.2). [a] Farming system resource of importance to threatened species: NH = nesting habitat, FH = foraging habitat, Gr = rice/cereal grain and Ca = animal carcasses. NB indicates that the dependence occurs in the non-breeding season only. [b] Threatened status: CR = Critically Endangered, EN = Endangered, VU = Vulnerable and NT = Near Threatened. [c] Extent of species' ranges, in the developing world only. [d] Species also occurs in developed countries.

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[1] Since publication of this paper, the author has become aware of two other species dependent on agricultural landscapes, the Ethiopian Bush-crow *Zavattariornis stresemanni* in pastoral systems of southern Ethiopia (Donald et al. 2012) and Tuamotu Kingfisher *Todiramphus gambieri* which selects coconut plantations with burnt understorey to feed on the Niau Atoll of French Polynesia (Coulombe et al. 2011).

In pastoral systems, domestic livestock may mimic or substitute crucial ecosystem functions once provided by wild herbivores, now extirpated or scarce. Consequently, many threatened species, such as larks and terrestrial waders, now appear reliant on livestock for maintaining habitat suitability in extensive savannas, rangelands and agro-forestry systems (Table 2.1.). Inappropriate exclusion of livestock from wetland or grassland systems can lead to declines of dependent biodiversity as has occurred in certain Indian conservation programmes (Lewis 2003). Domestic livestock can also be a vital food source for carcass-feeders such as South Asian *Gyps* and *Sarcogyps* vultures – so long as diclofenac residue is absent (Houston 1996; Pain et al. 2003).

Arable systems can provide abundant invertebrate prey, cereal grains and weed seeds, particularly in low-input cereal and rice farming. Species such as Asian Crested Ibis *Nipponia nippon* have benefited from long historical associations with traditional arable agriculture. Numerous crane species forage on agricultural land benefiting from spilt cereal grains (Table 2.1.), similar to the use of farmland by Common Crane *Grus grus* in Europe (Franco et al. 2000). In Asia, remaining areas of low-to-medium intensity rice cultivation provide stubbles that support wintering granivorous passerines, such as yellow-breasted bunting *Emberiza aureola*. Such production systems are now increasingly rare and threatened (Gray et al. 2007).

Mixed farming, combining pastoral and arable land-use within a landscape, is particularly important with its heterogeneity and small-scale complexity providing varied foraging resources and nest sites (van der Weijden et al. 2010). In Morocco, the Critically Endangered Northern Bald Ibis *Geronticus eremita* feeds in a mosaic of extensively grazed semi-arid littoral steppe and low-intensity, traditionally-cultivated barley fields and fallows (Bowden et al. 2008). Small-scale cultivation occurs close to, or amongst, littoral steppe habitat kept open by goats and sheep. This combination of pastoralism and crops create habitat conditions with a high density of invertebrate and lizard prey accessible to the ibis. Agricultural intensification associated with human population growth is threatening the long-term viability of this mixed farming system.

Numerous examples come from ancient, traditional farming systems, where species such as Asian Crested Ibis and Bengal Florican *Houbaropsis bengalensis* could shift to agricultural land uses over centuries or millennia, developing increasing dependency on these systems as their natural habitats were lost. Other cases have arisen much more recently where new land-use practices have replaced the key ecosystem processes that open-habitat species require. The Sociable Lapwing *Vanellus gregarius* (Fig. 2.2.) became reliant on agriculture

Figure 2.2. The Critically Endangered Sociable Lapwing *Vanellus gregarius* (a) which depends on grazed steppic grasslands (b) during the breeding season. Photographs courtesy of Maxim Koshkin.

in the twentieth century as the declining influence of native ungulates coincided with the creation of new rural livelihoods and novel farmed landscapes (Kamp et al. 2009). A large number of our cases of agricultural dependency come from Asia and Africa. This is perhaps related both to the ancient history of pastoralism and cereal agriculture in these regions, and to ecosystem functions now being carried out by livestock following recent extirpations of native ungulates.

These developing-world cases provide wider relevance for the semi-natural habitats paradigm. New and stronger dependencies are likely as agriculture continues to replace habitats and ecosystem processes in these countries. Developing-world farming systems may support a growing set of distinct taxa, although open-habitat species may still be lost where they occupy an ecological niche not substituted by human land-use, or where agricultural change is particularly severe. Further research is needed into the value of low-impact agriculture in the developing world, particularly for non-avian taxa, so that agriculture’s importance is better understood and valuable landscapes are identified.

2.6. Prospects for low-impact agriculture and associated biodiversity

Low-impact agriculture benefits a suite of threatened species in the developing world but is under threat from economic change. Escalating food prices create incentives for agricultural investment by new, external actors (Godfray et al. 2010) bringing infrastructure and high-input production methods that cause rapid land-use transition. The consequences of industrialised agriculture for greenhouse gas emissions and environmental problems (such as salinization, aquifer depletion and soil erosion), combined with increasing costs of inorganic fertilizers, may challenge the long-term viability of industrial agriculture (MEA 2005). However, economic drivers and the current failure of markets to capture externalities will probably sustain these models in the short to medium term. This represents an immediate threat to low-impact agriculture and could bring losses to open-habitat biodiversity.

Although large-scale industrial agriculture can benefit national economies and increase food production, it often threatens the livelihoods and social stability of rural communities (Cook 2009; MEA 2005). Corrupt institutions lacking transparency and accountability, weak land tenure and marginalised status can leave rural communities vulnerable to land concessions, land grabbing and mass-privatization of common resources (Cotula et al. 2009). These factors threaten wildlife-compatible pastoral economies in semi-natural grassland and savanna ecosystems causing conversion to ranching or cereal agriculture (Norton-Griffiths 1995). For example, land conversion in the Tonle Sap floodplain of Cambodia is eroding customary land rights and replacing pastoralism and traditional rice farming with intensive, irrigated rice cultivation, putting Bengal Florican at serious risk of extirpation (Gray et al. 2007).

Where high-input agriculture threatens both people and wildlife, conservation could attempt to halt, or at least delay, land-use transition by empowering rural communities. In such cases, supporting social justice and local land-use entitlement could provide a win-win scenario that advances the mutualistic goals of biodiversity and livelihood protection, enabling low-impact agriculture to persist, at least in the short term. Conservation goals may be particularly closely aligned with livelihoods when other opportunities are limited, such as in infertile marginal lands or hostile environments. However, economic changes and greater access to technology and markets will still encourage transition to more profitable, higher-yielding practices (Lambin et al. 2001), even where the impacts of external actors can be alleviated.

Small-scale farming is being championed as an alternative to industrial models. Characterised by low mechanical or chemical inputs with high crop complexity and high labour intensity, this form of agriculture could deliver greater productivity in relation to land area and provide a more sustainable means of future food supply (Perfecto & Vandermeer 2010). New models from both the development and conservation agendas propose that small-scale agriculture could achieve greater food production, food security, ecological and social resilience and poverty reduction (FAO 2007; IAASTD 2009), as well as promoting biodiversity conservation (Knoke et al. 2009; Perfecto & Vandermeer 2010). Nevertheless, achieving these socio-economic goals will require that existing small-scale farming systems are developed (Hazell et al. 2007), making the prospects for wildlife dependent on low-impact agriculture unclear. Intensification of production may prove detrimental to species dependent on extensive techniques, and with nearly one third of the human population living on small farms (Hazell et al. 2007) the impacts of agricultural development could be considerable.

2.7. The conservation response: applying the semi-natural habitats approach

The widespread transformation of low-impact agriculture appears likely, whether through extrinsic actors or internal agricultural development. Where threatened biodiversity is dependent on agriculture, minimising the threat of rapid industrialisation is a crucial first step. However, merely defending community entitlements to resist land-grabbing by external actors may not guarantee the status quo in the face of economic pressures and technological opportunities. Conservation should prepare for intervention, developing and adopting a range of policy mechanisms with the aim of maintaining, supporting or mimicking beneficial land management; thereby transferring the semi-natural habitats paradigm to the developing world. Effective interventions must be harmonised with socio-economic policies to ensure social development is not prevented (Adams et al. 2004). Curtailing economic growth or constraining livelihood opportunities could result in stakeholder discontent or threaten a policy’s long-term viability.

Various policy measures offer economic opportunities, incentives or development benefits to stakeholders, including: market enhancements such as certification schemes; community-based ecotourism; payments for environmental services; direct payments for conservation and conservation concessions (Bennett 2000; Ferraro & Kiss 2002). Such instruments could reward farmers for the take-up or continuation of valuable agricultural practices sustaining open-habitat species. Education to raise awareness of sustainable land management and resource use combined with disincentives for bad practice, such as enforcement of wildlife protection legislation and compulsory public disclosure of practices will also be important (Bruner et al. 2001). In many cases, policy measures would support rather than replace existing livelihoods, though compensation may be required for lost opportunities of developing higher-yielding, higher-impact agriculture. The need to quantify the costs of wildlife-friendly farming is recognised in the developed world (House et al. 2008), but elsewhere these costs and the necessary levels of compensation or incentive deserve further research.

Conservation of the Endangered Sharpe’s Longclaw *Macronyx sharpei* in Kenya is beginning to adopt the semi-natural habitat approach. The species requires short-sward grassland maintained by livestock, a habitat being lost to both agro-business- and smallholder-scale arable cultivation (Muchai et al. 2002). Land purchases are alleviating the threat of habitat conversion and grazing lets, administered by the community, provide income while creating suitable habitat conditions. At a larger scale, sheep-rearing is being advocated to provide a livelihood alternative and deliver habitat management for conservation (P. Matiku pers. comm.). Training and marketing is provided to encourage uptake of sheep-farming, supplemented by bee-keeping and an emerging eco-tourism scheme to provide further livelihood development.

Appropriate policy instruments will be highly context dependent. The pace of economic development, land entitlement of farmers, political transparency, complexity of stakeholder relations, strength of institutions and empowerment of local people are all important considerations (Salafsky et al. 2001). European policies such as agri-environmental schemes may be relevant in some instances, particularly where institutions are well-developed and legitimate. However, geographical transfer of policy measures will require assessments of their suitability under different social, political and economic conditions. The ability to undertake conservation of open-habitat species in agricultural landscapes will also depend on finding sustainable sources of funding – a challenge to be addressed for species conservation in general. Where financial resources are scarce and priorities have to be drawn, the decision on whether to conserve biodiversity in natural versus anthropogenic landscapes should be based on evidence regarding relative threat, conservation value, cost and likelihood of success.

2.8. Reconciling conservation and global food production

Protecting biodiversity in the face of projected rises in food demand is a challenge. To reconcile the aims of conservation and agricultural development, conservation has proposed a trade-off between two approaches: wildlife-friendly farming and land sparing (Ewers et al. 2009; Green et al. 2005; Phalan et al. 2011b). The former attempts to conserve species on farmland but with costs to yield, therefore requiring more land in cultivation. The latter would intensify agriculture to increase yields, reducing the need to convert further natural habitat to agriculture (Balmford et al. 2005). This trade-off can potentially be resolved using a model examining the response of species population density to agricultural yield. Where increases in yield cause steep (concave) declines in population density, land sparing through intensification is most appropriate as it gives greater regional species abundance for a given level of agricultural yield (Green et al. 2005; Phalan et al. 2011b).

However, this model assumes population density is always maximal in an existing and available natural habitat, with lower densities in all forms of agriculture and a monotonic decline with increasing yield [1]. This does not apply to those open-habitat species now dependent on agriculture, for which natural habitats or processes are absent and maximal density occurs along the gradient of human land-use and agricultural yield. While agricultural intensification, offset by land sparing, may be an appropriate strategy in frontier ecosystems (Sodhi et al. 2010), elsewhere it may heighten the risk of extinction for biota reliant on low-impact agriculture. This form of agriculture is the only option for such species, at least in the absence of large scale restoration of natural habitats and ecosystem function. A further limitation is the model's assumption that meeting human need depends solely on the volume of agricultural production. While markets may drive increased production, human welfare is often better served by resilient livelihoods, social security and adequate entitlements, all of which can be threatened by intensified industrial agriculture.

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[1] This statement is erroneous and this mistake was noticed after publication. The model does not make assumptions about where population density is maximal and considers a range of density-yield curves. Nevertheless, the majority of species so far used to illustrate the model have demonstrated maximal density in an existing available natural habitat, which does not apply to open-habitat species. The implications of this are discussed further in section 2.9.4 Appendix D.

Conservation strategies that provide not just for forest species, but also agriculture-dependent species, will require a mixture of intensification, land sparing and extensive low-impact agriculture that should be optimised for any particular region (Fischer et al. 2008). Agriculture's paradoxical nature, as both a great threat to biodiversity and a valuable land-use that sustains open-habitat species (van der Weijden et al. 2010), would be better represented by such a compromise. We have shown that agricultural dependency is widespread across the developing world; however, uncertainty remains as to its relative frequency and regional variability. An urgent task for conservation is to identify the land-use practices and anthropogenic landscapes important to biodiversity and to develop the mechanisms to maintain them before they are lost through land-use change.

2.9. Appendices

2.9.1. Appendix A: Details of search terms for the analysis of bird species habitat associations using the *IUCN Red List for birds* database.

We systematically examined the habitat associations of globally threatened birds species listed in the *IUCN Red List for birds* database (BirdLife International 2011) using search terms available within BirdLife International's online Data Zone resource. Terms for region included Europe, North America, Australasia, Asia, Africa and South America, with the latter three combined to create overall statistics for the developing world. Habitat terms were "forest", "grasslands", "savannas" and "terrestrial artificial landscapes".

No search terms for agricultural land-uses were available in the Data Zone resource, therefore we used terrestrial artificial landscapes as a proxy that combines agricultural land, plantations and rural gardens, but may also include urban areas as well as former subtropical/tropical forest lands heavily-degraded by combinations of logging, pastoralism, swidden agriculture and collection of fuelwood and construction timber by local communities. The considerably greater global land area covered by crops and grazing pastures (48%) compared to settlement (1%; Erb et al. 2007) justifies this application. While we acknowledge that some associations with terrestrial artificial landscapes will not represent use of agricultural areas, such errors are likely to be minor.

Bird species of grassland and savanna were considered in our analysis because of the importance of human land use, particularly agriculture, in these ecosystems. Above precipitation thresholds savannas are unstable systems maintained free from woodland cover by fire and herbivory (Sankaran et al. 2005). Though savannas in low rainfall regions may be more stable states, degree of woody cover is nevertheless affected by fire and grazing. Human activity may threaten some of these climatically limited grasslands and savannas (Krapovickas & Giacomo 1998) but it is increasingly recognised that many have been modified by historical human land use or are of entirely anthropogenic origin (White et al. 2000). Examples of anthropogenic savannas, converted from forest by human use of fire, include those of the Indian sub-continent, tropical Australia and New Guinea, North America and the Mediterranean (Keeley 2002; Lunt et al. 2006; Naveh 2007; Saha 2003). Human influenced savannas in Africa have been influenced by fire and pastoralism for millennia (Smith 1992). Elsewhere, humans have extended the altitudinal range of montane grasslands through forest clearance and livestock grazing (Bredenkamp et al. 2002) or have replaced the role of native ungulates in semi-natural grassland or savanna ecosystems.

2.9.2. Appendix B: Primary literature sources for species demonstrating a dependency on low-impact agriculture in the developing world (Table 2.1.), listed by species in alphabetical order of common name.

Asian Crested Ibis *Nipponia nippon*

Li, X., Li, D. (1998). Current state and the future of the crested ibis (*Nipponia nippon*): a case study by population viability analysis. *Ecological Research* 13: 323-333.

van der Weijden, W., Terwan, P., Guldemond, A. (2010). Farmland Birds Across the World. Lynx Edicions, Barcelona.

Bengal Florican *Houbaropsis bengalensis*

Gray, T.N.E., Chamnan, H., Collar, N.J., Dolman, P.M. (2009). Sex-specific habitat use by a lekking bustard: conservation implications for the critically endangered Bengal Florican (*Houbaropsis bengalensis*) in an intensifying agroecosystem. *Auk* 126: 112-122.

Gray, T.N.E., Collar, N.J., Davidson, P.J.A., Dolman, P.M., Evans, T.D., Fox, H.N., Hong, C., Ro, B., Seng, K.H., van Zalinge, R.N. (2009). Distribution, status and conservation of the Bengal Florican *Houbaropsis bengalensis* in Cambodia. *Bird Conservational International* 19: 1-14.

Black-necked Crane *Grus nigricollis*

Bishop, M.A. (1996). Black-necked crane (*Grus nigricollis*). Pages 184-194 in C.D. Meine, and G.W. Archibald, eds. The Cranes: Status Survey and Conservation Action Plan IUCN, Gland, Switzerland.

Blue Bustard *Eupodotis caerulescens*

Moreira, F. (2004). Distribution patterns and conservation status of four bustard species (family Otididae) in a montane grassland of South Africa. *Biological Conservation* 118: 91-100.

Blue Crane *Grus paradisea*

Allan, D.G. (1995). Habitat selection by Blue Cranes in the Western Cape Province and the Karoo. *South African Journal of Wildlife Research* 25: 90-97.

Botha's Lark *Spizocorys fringillaris*

BirdLife International (2011) IUCN Red List for birds. Available from http://www.birdlife.org. Accessed 25/11/2011.

Maphisa, D.H., Donald, P.F., Buchanan, G.M., Ryan, P.G. (2009). Habitat use, distribution and breeding ecology of the globally threatened Rudd's Lark and Botha's Lark in eastern South Africa. *Ostrich* 80: 19-28.

Buff-breasted Sandpiper *Tryngites subruficollis*

Lanctot, R.B., Blanco, D.E., Dias, R.A., Isacch, J.P., Gill, V.A., Almeida, J.B., Delhey, K., Petracci, P.F., Bencke, G.A., Balbueno, R.A. (2002). Conservation status of the Buff-breasted Sandpiper: historic and contemporary distribution and abundance in South America. *Wilson Bulletin* 114: 44-72.

Dupont's Lark *Chersophilus duponti*

Seoane, J., Justribó, J.H., García, F.R., J., Rabadán, C., Atienza, J.C. (2006). Habitat-suitability modelling to assess the effects of land-use changes on Dupont's lark *Chersophilus duponti*: a case study in the Layna Important Bird Area. *Biological Conservation* 128: 241-252.

Great Bustard *Otis tarda*

Hildago de Trucios, S.J. (1990). World status of the Great Bustard (*Otis tarda*) with special attention to the Iberian peninsula populations. *Miscellania zoologica* 14: 167-180.

Great Indian Bustard *Ardeotis nigriceps*

Dutta, S., Rahmani, A.R., Jhala, Y.V. (2011). Running out of time? The great Indian bustard *Ardeotis nigriceps* - status, viability, and conservation strategies. *European Journal of Wildlife Research* 57: 615-625.

Grey Crowned-crane *Balearica regulorum*

Meine, C.D., Archibald, G.W. (1996). The Cranes: Status Survey and Conservation Action Plan. IUCN, Gland, Switzerland.

Hooded Crane *Grus monacha*

See Grey Crowned-crane.

Indian Vulture *Gyps indicus*

Houston, D.C. (1996). The Effect of Altered Environments on Vultures. Pages 327-336 in D. Bird, D. Varland, and J. Negro, eds. Raptors in Human Landscapes: Adaptations to Built and Cultivated Environments. Academic Press, London.

Pain, D.J., Cunningham, A.A., Donald, P.F., Duckworth, J.W., Houston, D.C., Katzner, T., Parry-Jones, J., Poole, C., Prakash, V., Round, P.D., Timmins, R. (2003). Causes and effects of temporospatial declines of *Gyps* vultures in Asia. *Conservation Biology* 17: 661-671.

Jerdon’s Courser *Rhinoptilus bitorquatus*

Jeganathan, P., Green, R.E., Norris, K., Vogiatzakis, I.N., Bartsch, A., Wotton, S.R., Bowden, C.G.R., Griffiths, G.H., Pain, D.J., Rahmani, A.R. (2004). Modelling habitat selection and distribution of the critically endangered Jerdon’s courser *Rhinoptilus bitorquatus* in scrub jungle: an application of a new tracking method. *Journal of Applied Ecology* 41: 224-237.

Liben Lark *Heteromirafra sidamoensis*

Donald, P.F., Buchanan, G.M., Collar, N.J., Dellelegn Abebe, Y., Gabremichael, M.N., Mwangi, M.A.K., Ndang'ang'a, P.K., Spottiswoode, C.N., Wondafrash, M. (2010). Rapid declines in habitat quality and population size of the Liben (Sidamo) Lark *Heteromirafra sidamoensis* necessitate immediate conservation action. *Bird Conservation International* 20: 1-12.

Spottiswoode, C.N., Wondafrash, M., Gabremichael, M.N., Dellelegn Abebe, Y., Mwangi, M.A.K., Collar, N.J., Dolman, P.M. (2009). Rangeland degradation is poised to cause Africa’s first recorded avian extinction. *Animal Conservation* 12: 249-257.

Northern Bald Ibis *Geronticus eremita*

Bowden, C.G.R., Smith, K.W., El Bekkay, M., Oubrou, W., Aghnaj, A., Jimenez-Armesto, M. (2008). Contribution of research to conservation action for the northern bald ibis *Geronticus eremita* in Morocco. *Bird Conservational International* 18: S74-S90.

Pale-headed Brush-finch *Atlapetes pallidiceps*

Oppel, S., Schaefer, H.M., Schmidt, V., Schröder, B. (2004). Habitat selection by the pale-headed brush-finch (*Atlapetes pallidiceps*) in southern Ecuador: implications for conservation. *Biological Conservation* 118: 33-40.

Red-headed Vulture *Sarcogyps calvus*

See Indian Vulture.

Rudd’s Lark *Heteromirafra ruddi*

Maphisa, D.H., Donald, P.F., Buchanan, G.M., Ryan, P.G. (2009). Habitat use, distribution and breeding ecology of the globally threatened Rudd’s Lark and Botha’s Lark in eastern South Africa. *Ostrich* 80: 19-28.

Saffron-cowled Blackbird *Xanthopsar flavus*

Petry, M.V., Krüger, L. (2010). Frequent use of burned grasslands by the vulnerable Saffron-Cowled Blackbird *Xanthopsar flavus*: implications for the conservation of the species. *Journal of Ornithology* 151: 599-605.

Sharpe’s Longclaw *Macronyx sharpei*

Muchai, M., Lens, L., Bennun, L. (2002). Habitat selection and conservation of Sharpe's longclaw (*Macronyx sharpei*), a threatened Kenyan grassland endemic. *Biological Conservation* 105: 271-277.

Sierra Madre Sparrow *Xenospiza baileyi*

Cabrera-Garcia, L., Montes, J.A.V., Weinmann, M.E.E. (2006). Identification of priority habitats for conservation of the Sierra Madre sparrow *Xenospiza baileyi* in Mexico *Oryx* 40: 211-217.

Slender-billed Vulture *Gyps tenuirostris*

See Indian Vulture.

Sociable Lapwing *Vanellus gregarius*

del Hoyo, J., Elliot, A., Sargatal, J. (1996). Handbook of the Birds of the World, Volume 3: Hoatzin to Auks. Lynx Edicions, Barcelona.

Kamp, J., Sheldon, R.D., Koshkin, M.A., Donald, P.F., Biedermann, R. (2009). Post-Soviet steppe management causes pronounced synanthropy in the globally threatened sociable lapwing *Vanellus gregarius*. *Ibis* 151: 452-463.

Southern Bald Ibis *Geronticus calvus*

Manry, D.E. (1985). Distribution, abundance and conservation of the bald ibis *Geronticus calvus* in Southern Africa. *Biological Conservation* 33: 351-362.

St Helena Plover *Charadrius sanctaehelenae*

McCulloch, N. (2009). Recent decline of the St Helena Wirebird *Charadrius sanctaehelenae*. *Bird Conservation International* 19: 33-48.

White-rumped Vulture *Gyps bengalensis*

See Indian Vulture.

White-shouldered Ibis *Pseudibis davisoni* [1]

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[1] Chapters 5–6 now provide a detailed demonstration and discussion of the White-shouldered Ibis’ likely dependence on farming.

Wright, H.L., Buckingham, D.L., Dolman, P.M. (2010). Dry season habitat use by critically endangered white-shouldered ibis in northern Cambodia. *Animal Conservation* 13: 71-79.

Wright, H.L., Vorsak, B., Collar, N.J., Gray, T.N.E., Lake, I.R., Phearun, S., Rainey, H.J., Vann, R., Ko, S., Dolman, P.M. (2010). Establishing a national monitoring programme for White-shouldered Ibis in Cambodia. *Ibis* 152: 206-208.[1]

Yellow-breasted Bunting *Emberiza aureola*

BirdLife International (2011) IUCN Red List for birds. Available from http://www.birdlife.org. Accessed 25/11/2011.

2.9.3. Appendix C: Response to this paper by Phalan et al. (2012), entitled *Agriculture as a key element for conservation: reasons for caution* (reproduced here with permission from Ben Phalan) and presented as published in *Conservation Letters*; this is not the work of the author.

We agree with Wright et al. (2012a; [chapter 2]) that it is important to consider species of open habitats when assessing the impact of agricultural policy on landscapes where such species occur. However, there are at least four reasons why conservationists should be cautious about the idea that agriculture is a key element for conservation in the developing world (or indeed anywhere):

(1) Observing that most individuals of some bird species make use of agricultural habitats at some stage of their life history is insufficient to tell us whether preserving those habitats is desirable for the long-term conservation of other biodiversity, of all birds or even of those species themselves. All species have survived without agriculture for most of their evolutionary history. Most species which are now found largely on agricultural land use non-agricultural habitats as well, including open natural and semi-natural habitats. The methods we implemented in a recent analysis (Phalan et al. 2011b) assess the proportion of species which would benefit most from maximising the area of low-yielding agriculture, maximising the area of natural habitat by producing the same quantity or value of agricultural goods from a smaller area of high-yielding agriculture, or an intermediate strategy. Our approach depends upon measurements of population density across a range of land uses (and not, as Wright et al. incorrectly state, an assumption that "population density is always maximal in an existing and available natural habitat, with lower densities in all

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[1] This project report is another output of this thesis's research, published in *Ibis* to present preliminary results and update on research progress.

forms of agriculture and a monotonic decline with increasing yield”). The paper by Wright et al. does not present any such measurements.

(2) Decisions about land use have off-site consequences (Phalan et al. 2011a). There might be landscapes where data suggest the best way to conserve certain species is to attempt to “fossilise” some low-yielding farming practices. However, sparing low-yielding farmland in the face of rapidly rising demand for farm products would require us to accept agricultural expansion or yield increases elsewhere, with impacts on other species. Our approach offers a method to quantify those leakage effects on particular species, and on wider groups of species. Some of these other groups may have an even smaller proportion of species that tolerate agriculture than do birds. For example, low levels of cattle grazing might maintain open habitats suitable for some birds, but might not be compatible with the conservation of the native herbivores that previously created such conditions.

(3) Intervention to keep constant those farming practices in low-yielding agricultural landscapes that allow birds to live in them is difficult. Species with most individuals currently living on agricultural land are at risk from future changes in agricultural technology and the demand for different crops. Of the bird species identified by Wright et al. as being “dependent on low-impact agriculture,” many are in fact threatened by changes in small-scale agriculture, and not just by large-scale “industrial” agriculture. Liben Lark *Heteromirafra sidamoensis* is an example, where relatively small changes in farming practices by local people have taken the species close to extinction (Donald et al. 2010).

(4) There is an alternative to being constrained by current patterns of land use: habitat restoration might be an effective way of conserving some species in landscapes where most or all natural habitats have been converted. Once again, expanding or re-creating areas of natural habitat will be practical only if increasing production elsewhere reduces demand for farmed land. Density-yield analyses of the type we advocate would help to clarify whether such a restoration-based approach might be appropriate, not just for a handful of bird species but for a broader sweep of the regional biota.

2.9.4. Appendix D: Response to Phalan et al. (2012), entitled *Low-impact agriculture requires urgent attention not greater caution: response to Phalan and colleagues* and presented as published in *Conservation Letters* (Wright et al. 2012b).

Phalan et al. (2012) set out to present four reasons for caution when considering agriculture in developing-world conservation. However, contrary to their suggestion, our paper (Wright et al. 2012a; [chapter 2]) emphasised not those species that make use of agricultural habitats at some stage of their life history, but those whose populations depend on agriculture. We agree that the agricultural dependence of birds does not indicate that other biodiversity will follow the same pattern; although birds are widely used as proxies, their congruence is incomplete and scale-dependent (Prendergast et al. 1993; Schulze et al. 2004). We know from Europe that distinctive and highly valued assemblages of plants and invertebrates now depend on semi-natural habitats created by traditional farming practices that substituted for lost natural processes. Large mammals are unlikely to show such a response. Crucially however, the loss of large herbivores and extirpation of ecosystem functions they provided often resulted in the dependency of open-habitat birds on agricultural practices, where these mimicked the processes that species require. While it is obvious that species evolved prior to the advent of agricultural transformation, this is not useful when their natural habitats are now absent. Like Phalan et al., we also advocate restoration of large scale natural ecosystem dynamics, but this is not immediately practical in many regions. Not preserving species in the semi-natural and farmed habitats in which they occur risks their loss in the short to medium term. The paradigm of semi-natural habitats, essential to European conservation, therefore has wide application in developing countries also.

Phalan et al. are correct to point out that, within the constraints of finite land resources and increasing demands for food, any action to conserve one set of species must be traded off against other biodiversity. Their trade-offs model (Green et al. 2005; Phalan et al. 2011a) provides an appropriate starting point to examine the optimal balance of land-sparing versus land-sharing. We urge the wider adoption of such models in regions of contrasting biota and land-use history for which different trade-offs may apply. At least some degree of land-sparing may be crucial in frontier regions where the emphasis is on conservation of forest biota. In contrast, responses to anthropogenic land use may differ in grassland and savanna biomes and landscapes with a long history of human transformation, particularly in systems of extensive pastoralism and traditional cereal cultivation. The choice between land-sparing and wildlife friendly farming should not be simplified into a dichotomy; a mixed approach may conserve the broadest range of a region’s biota, especially in regions with contrasting habitats. Furthermore, strategies must also account for a range of other, often context-specific, social, political and ecological considerations that the simple trade-offs model does not yet incorporate (Phalan et al. 2011a; Phalan et al. 2011c); for example the size, range and conservation significance of individual species’ populations. Advocating a single strategy may therefore be unhelpful, particularly beyond the regional scale.

Contrary to the suggestion by Phalan et al., we have not advocated “fossilising” low-impact farming practices. Rather, we made clear (Wright et al. 2012a; [chapter 2]) that threats of land-use transformation come not just from the land-grabbing of external actors, but also from within rural communities. Conservation must design mechanisms that are compatible with social and economic change, not defend uneconomic agricultural systems. The conservation imperative therefore, is to urgently identify those cases where agriculture currently sustains valuable biodiversity, and to develop instruments to maintain or mimic such land use while supporting development (see also Fischer et al. 2012). Our paper highlighted both the threat to farming practices valuable to agriculture-dependent species and the challenges in maintaining them; these are reasons why conservation should pay urgent attention to beneficial farming systems, not reasons for caution.

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Chapter 3

First census of the White-shouldered Ibis *Pseudibis davisoni* reveals roost-site mismatch

White-shouldered Ibises *Pseudibis davisoni* at a wet season roost.

Presented here as published for the short communication:

Wright, H.L., Collar, N.J., Lake, I.R., Net Norin, Rours Vann, Sok Ko, Sum Phearun, Dolman, P.M. (2012). First census of white-shouldered ibis *Pseudibis davisoni* reveals roost-site mismatch with Cambodia’s protected areas. *Oryx* 46: 236-239.

3.1. Abstract

The population size of the Critically Endangered White-shouldered Ibis *Pseudibis davisoni* has always been poorly known. The first-ever census across Cambodia in 2009–2010 using simultaneous counts at multiple roost sites found substantially more birds than previously estimated, with a minimum of 523 individuals. The census allowed a revised global population estimate of 731–856 individuals, increasing hope for the species' long-term future. However, the largest subpopulations are imminently threatened by development and c. 75% of the birds counted in Cambodia occurred outside protected areas.

3.2. Introduction and methods

Cambodia is a stronghold for threatened biota in South-East Asia (CEPF 2007), with a protected-area system covering 31% of the country's land area (FAO 2010). However, with status and distributional data lacking for much of Cambodia's wildlife (Neou 2004), protected areas may not provide adequate coverage for certain key species, a problem frequently found in other parts of the world (Beresford et al. 2010; Brooks et al. 2004; Rodrigues et al. 2004). The first-ever census of the Critically Endangered White-shouldered Ibis *Pseudibis davisoni* reveals a new instance of this issue.

Considered one of the most threatened waterbirds in Indochina (BirdLife International 2001), the White-shouldered Ibis was widespread in the region until the 20th century. Now confined to Cambodia and adjacent southern Laos, plus one river in Indonesian Borneo (BirdLife International 2011), in 2000 it was assigned Critically Endangered status, with the population assessed at only 330 mature individuals in 2010 (BirdLife International 2011). Four major subpopulations were identified in Cambodia as biodiversity surveys achieved greater coverage of the country (BirdLife International 2006; Timmins 2008). However, population assessments remained informal, with no previous coordinated censuses.

In 2009 we implemented the first coordinated census to quantify subpopulations throughout Cambodia. White-shouldered Ibises are solitary dry-season breeders, becoming gregarious in the wet season (May–October) when they roost together in trees. Simultaneous wet-season roost counts can therefore provide minimum population numbers. We located roosts in and around the four sites known to hold most ibises: Kulen Promtep Wildlife Sanctuary, Lomphat Wildlife Sanctuary, the central section of the Mekong River [1] between Kratie and

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[1] Since publication, this site has been renamed the Mekong Flooded Forest.

Stung Treng towns, and Western Siem Pang Important Bird Area (Fig. 3.1.). Although complete coverage of these sites (totalling more than 960,800 ha) could not be achieved, all locations known or believed to hold important numbers of ibises were prioritised and intensively searched with assistance from local informants.

Nine coordinated counts took place over July–December 2009 and July–October 2010 at approximately monthly intervals. Observers remained a suitable distance from roosts to avoid disturbance, although this prevented distinguishing immatures from adults.

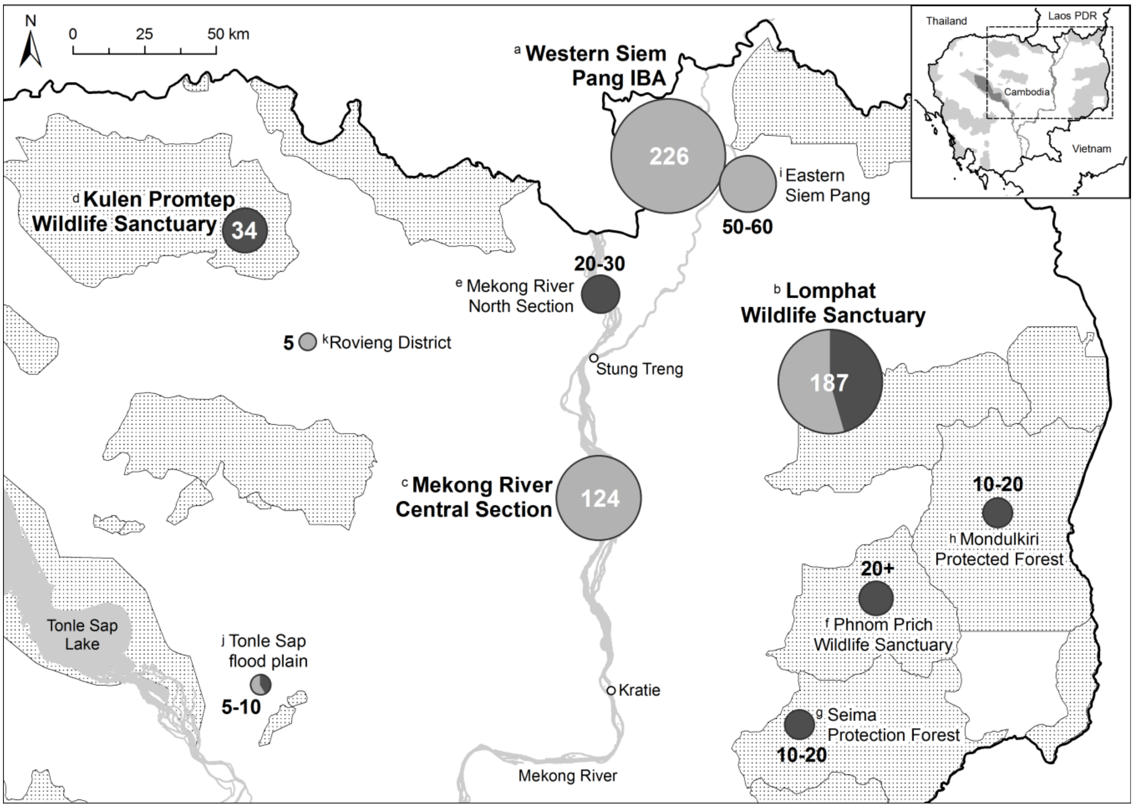

Figure 3.1. Known White-shouldered Ibis subpopulations in Cambodia and their recorded/estimated size. Pie-charts are scaled to population size and show percent of birds occurring within protected (dark grey) and unprotected (pale grey) areas. Figures inside pie-charts were obtained from roost counts, figures beside pie-charts are estimates. Stippled polygons represent protected areas. [a] Roost count September 2010; [b] roost count October 2010; [c] roost count October 2010[1]; [d] roost count September 2010; [e] Timmins (2006); [f,h] T. Gray pers. comm. 2011; [g] Bird et al. (2007); [i] HLW unpublished data; [j,k] Wildlife Conservation Society unpublished data.

Knowledge of roost locations steadily improved (from 18 in July 2009 to 39 in October 2010), making the counts increasingly comprehensive.

3.3. Results and discussion

The largest total count was 523 in October 2010, the final census; however, few additional birds were accumulated in the final three censuses. Despite only including four areas, this new minimum number exceeds the global population estimates of 330 mature individuals (BirdLife International 2011) and < 500 individuals of all ages (Timmins 2008). However, 74% of the ibises were at roosts outside the boundaries of protected areas. Western Siem Pang and the Mekong River central section, currently unprotected, together accounted for 58% of the ibises censused. At Lomphat 46% of birds were at roosts outside the demarcated Sanctuary.

Peak site counts provide a preliminary indication of the relative size of the four subpopulations (Fig. 3.1.): 226 birds in Western Siem Pang, 187 at Lomphat, 124 on the Mekong River central section and 34 at Kulen Promtep. Although these peaks were not obtained simultaneously, all were in September–October 2010. The minimum distance between sites is 47 km, and as yet we have no evidence that the ibises move this far. Count fluctuations within sites appeared unrelated to counts at other sites, being attributable instead to short-distance movements and changes in favoured roosts.

Combining roost census data with estimates for other, smaller populations, we propose that there is a minimum Cambodian population of 691–736 (Fig. 3.1.). Other populations have not yet been counted accurately but available data (for the Mekong River north section) and expert judgement (for Eastern Siem Pang, Mondulkiri Protected Forest, Rovieng district, Phnom Prich Wildlife Sanctuary, Seima Protection Forest and Tonle Sap floodplain) allow estimates to be made.

Given the increasingly comprehensive coverage of biodiversity inventories it is unlikely that further large subpopulations will be discovered in Cambodia. However, fuller coverage of the Mekong River central section could potentially reveal 100-200 more birds (R.J. Timmins pers. comm. 2011) and diffuse, lower-density populations may remain undetected elsewhere, particularly in Mondulkiri and Ratanakiri provinces. These putative populations are not included in our estimates.

The Indonesian population has been estimated at only 30–100 (BirdLife International 2011) and is decreasing (Meijaard et al. 2005); with no recent surveys its current size is uncertain. Although no records have come from Laos since the 1990s the proximity of Western Siem Pang birds to Xe Pian National Protected Area (across the border), plus minimal survey effort at the latter, suggests small numbers could still persist. We provisionally estimate there are 10–20 ibises in Laos. In Vietnam sightings have gone from few to almost none since the 1990s (R. Craik pers. comm. 2011). The species is now probably only a non-breeding visitor from Cambodia and thus Vietnam does not contribute to our figures. We therefore estimate a minimum global population of 731–856 birds [1].

This total is larger than previously estimated, providing hope that the long-term future of the species can still be secured. Nevertheless, these increased numbers reflect improved coverage and rigour of surveys rather than population recovery, and the species is still greatly threatened. Only 25.9–28.4% of White-shouldered Ibises occur in legally protected areas in Cambodia; globally the proportion is 25.9–26.8%. Western Siem Pang, currently the most important site, is unprotected and plantations will convert the majority of habitat by 2020 (BirdLife International 2010). Proposed dams threaten Lomphat Wildlife Sanctuary (BirdLife International 2010) and the unprotected Mekong River central section, which also faces encroaching human settlement (Timmins 2008). With 63–73% of the known global population in three imminently threatened sites and probably continuing declines in other areas, the White-shouldered Ibis is still Critically Endangered. Securing the Western Siem Pang and central Mekong subpopulations is now essential.

Given the large proportion of Cambodia already in reserves, the distributional mismatch between White-shouldered Ibis roosts and the country’s protected-area system is unfortunate. Protected area designations have prioritised the least impacted habitats furthest from settlement, whereas the ibis requires human-influenced habitats, feeding in seasonal pools grazed by livestock and in traditional agricultural fallows (Wright et al. 2010a; Wright et al. 2010b; [chapter 5]). If establishment of further protected areas is not possible, then special management zones (integrating human use and biodiversity protection) and conservation concessions, not yet attempted in Cambodia, could provide alternatives.

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[1] Coordinated roost-counts have continued and expanded since this paper’s publication, providing greater site coverage and finding more birds. Unpublished data from 2011 now suggest a minimum Cambodian population of 644 birds, and a likely minimum global population of 827–952 birds. A publication presenting updated estimates is planned for late 2012, following a fourth year of censuses.

However, the White-shouldered Ibis is not the only threatened species inadequately served by Cambodia's protected areas. The Critically Endangered *Gyps* and *Sarcogyps* vultures and Bengal Florican *Houbaropsis bengalensis* also have close association with humans through traditionally managed habitat and resources (Gray et al. 2009; Houston 1996). Such habitats and resources will be the first to deteriorate with economic development, putting these species at great risk of extinction. With the intensification of agriculture and expansion of cash-crop cultivation already a major threat in Cambodia, as across much of Indochina (CEPF 2007), such species urgently need attention to ensure their survival.

3.4. References

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Chapter 4

Foraging ecology helps resolve the paradox of a waterbird breeding in the dry season

White-shouldered Ibis *Pseudibis davisoni* foraging in dry cracked substrate at a *trapaeng* (waterhole).

4.1. Abstract

In contrast to the great majority of waterbirds in central Indochinese dry forests, and counterintuitively, White-shouldered Ibis *Pseudibis davisoni* nests in the mid- to late dry season, when water is scarce. To understand how this species successfully feeds and provisions chicks at the driest time of year, its diet and habitat use, and the habitat conditions influencing intake rate and prey density were studied. Ibis foraging observations, prey sampling and landscape-scale assessment of habitat availability were undertaken over two breeding seasons at seven, 49 and 58 waterholes respectively. White-shouldered Ibis avoided foraging in water but used all exposed substrates at waterholes, feeding on amphibians and small invertebrates. Amphibians were the most abundant prey type in waterhole substrates and accounted for 81% of overall prey intake. Amphibian prey intake rates and biomass density were greater in dry than moist and/or saturated substrates. Dry substrate was also the most widespread habitat type at waterholes in the peak dry season. By utilising the commonest dry season habitat and prey, and probing into substrate cracks and holes that other birds cannot readily reach, the ibis may achieve greater feeding success than other dry forest waterbirds, allowing it to breed when water is scarce. Estimated prey depletion was non-trivial, as a breeding pair requires nearly twice the amphibian biomass density at a medium-sized waterhole during the nesting period. Each pair therefore probably requires multiple waterholes, making landscape-scale habitat protection a necessity, and human harvesting of amphibians could threaten ibis if the offtake and spatial extent of collection increases.

4.2. Introduction

Rainfall and wetland hydrology exert important influences on waterbird foraging strategies and reproduction (Bildstein et al. 1990; Frederick et al. 2009; Kushlan 1986), as seasonally fluctuating water-levels create variable food abundance and availability (Frederick & Collopy 1989; Gawlik 2002). Reproductive responses to hydrological regimes may vary between ecosystems: waterbirds in arid environments may adopt flexible nesting cycles, coinciding with floodwaters that improve food abundance in otherwise resource-scarce environments (Halse & Jaensch 1989; Kingsford & Norman 2002). Elsewhere, the breeding season may correspond with water drawdown, as prey becomes more concentrated and accessible to feeding birds at receding pools (Frederick & Collopy 1989; Russell et al. 2002). Responses also vary between waterbird species, with different habitats or prey requirements resulting in contrasting nest success and breeding phenology between members of the same family (e.g. egrets; Maddock & Baxter 1991) or waterbirds at a shared wetland (Berruti 1983).

The tropical dry forests of central Indochina support a distinctive assemblage of large-bodied waterbird species, including two ibises, two adjutants, two other storks and one crane (of which five are listed as threatened on the IUCN Red List; BirdLife International 2012a). Many of these species rely on widely distributed seasonal waterholes, where water-levels fluctuate with the monsoonal climate, exposing substrate as pools diminish in the dry season (November–April). As in many wetlands, the breeding phenology of dry forest waterbirds appears closely tied to water availability, as nesting mostly takes place during the wet season (May–October) or from the late wet to mid-dry seasons (c. September–February; Clements et al. in press), when pools remain full or largely flooded. However, the White-shouldered Ibis *Pseudibis davisoni* (a solitary breeder) is a notable exception, nesting in the mid- to late dry season (Fig. 4.1.), when water is at its scarcest and many waterholes completely dry out (Wright et al. 2010). This intriguing reversal of the normal breeding pattern, contrasting strongly with that of the sympatric Giant Ibis *Thaumatibis gigantea* (Keo 2008), indicates that the White-shouldered Ibis must successfully feed and provision chicks in water-scarce conditions, but how it does so is not yet known.

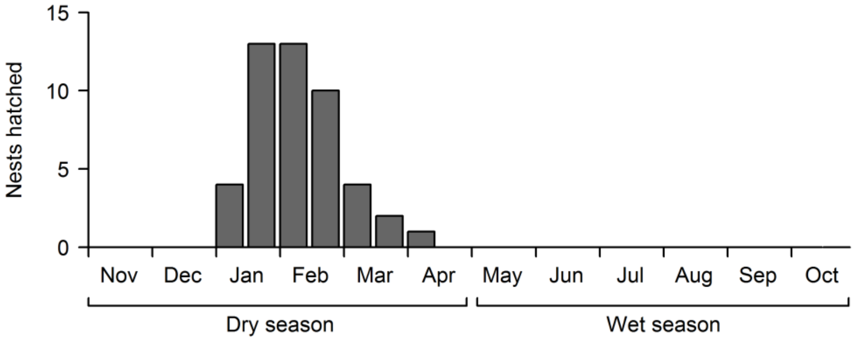

Figure 4.1. Frequency of White-shouldered Ibis nests hatched by half-month period in three breeding seasons from 2008–2011 (*n* = 47) in Western Siem Pang IBA, a dry forest landscape in northern Cambodia. Incubation (taking 30.7 ± 2.7 days, mean ± SD, *n* = 17) begins as early as mid-December with chick provisioning during January–May (lasting a further 38.7 ± 6.6 days until fledging, *n* = 22). Nest trees were located through active searching by field staff and reports from local people, and were monitored (by ground-based observation) every 5–7 days to check nest status.

The White-shouldered Ibis (listed as Critically Endangered by IUCN) underwent a severe decline in the twentieth century, becoming largely confined to dry forests and river channels of Cambodia, where 85–95% of the world's remaining 731–856 birds now occur (Wright et al. 2012b; chapter 3). Agricultural plantations and infrastructural developments are now the biggest threats to its survival (BirdLife International 2012b), but its dry forest foraging habitats, including waterholes, are likely to be sustained by local land management practices such as grazing by livestock (chapter 6). Studies of broad-scale habitat selection show that waterholes ("*trapaengs*"), particularly those with low vegetation, are key breeding-season habitats for foraging ibis (Wright et al. 2010; Wright et al. 2012a; chapter 5).

But how do foraging White-shouldered Ibis profit from *trapaengs* at the driest time of year? To explain this paradox it was predicted that: (1) ibis diet would comprise the most abundant prey types in this season; (2) as a dry-season breeder, the ibis would forage successfully in a range of substrate types increasingly exposed by water drawdown at *trapaengs* (rather than relying on diminishing pools of water); and (3) ibises would forage less successfully where tall, dense vegetation restricted access to the ground. These predictions were tested with measures of ibis habitat use, prey selection, intake rate and prey biomass density, and estimated prey depletion at *trapaengs* during the ibis breeding season. The likely influence of *trapaeng* hydrology and prey scarcity on ibis breeding in the mid- to late dry season is discussed, comparing the species's ecology to that of other dry forest waterbirds and considering implications for ibis conservation.

4.3. Methods

4.3.1. Study area

Ibis foraging ecology was studied within Western Siem Pang Important Bird Area (IBA), Stung Treng province, northern Cambodia (14°07'N 106°14'E, Fig. 4.2.). This 138,000 ha site contains the largest known subpopulation of White-shouldered Ibis, a minimum of 226 birds (Wright et al. 2012b; chapter 3), and comprises deciduous dipterocarp forest, a savannah-like woodland with an open and annually burnt grassy understorey. The forest is interspersed with patches of agriculture, grassland and mixed deciduous and semi-evergreen forests. Rainfall is strongly seasonal, with monthly means as high as 333 mm and as low as 0.9 mm in the wet and dry seasons respectively (Thuon & Chambers 2006). *Trapaengs* (0.001–3.4 ha) occur frequently and water-levels, vegetation and habitat heterogeneity vary

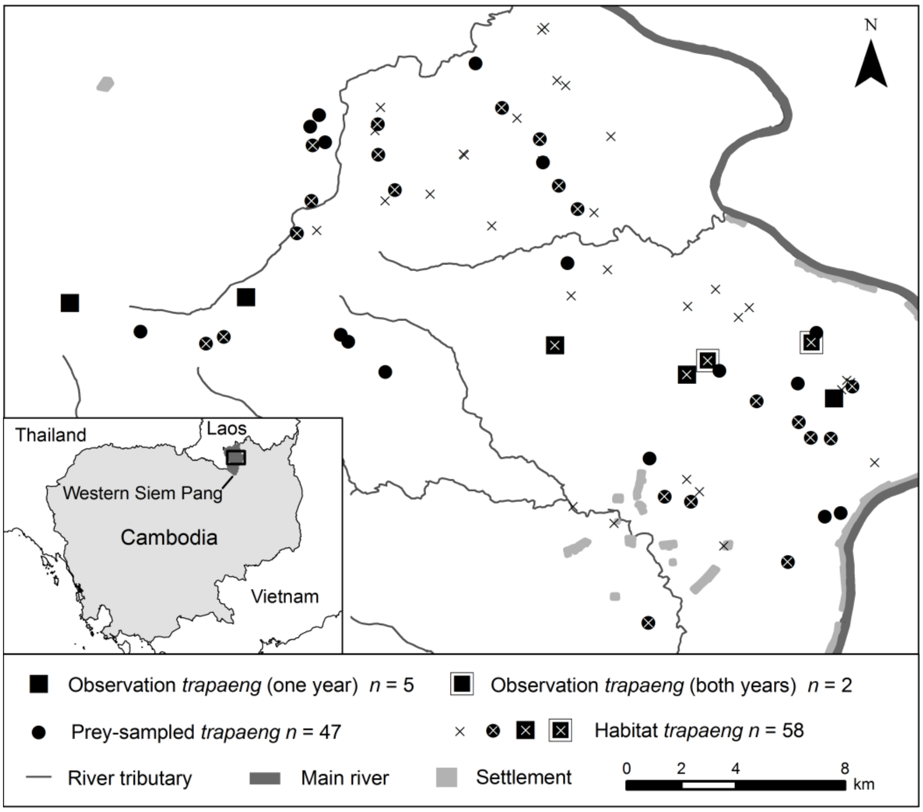

Figure 4.2. Map of the study area in Western Siem Pang Important Bird Area (IBA), illustrating the distribution of three *trapaeng* samples surveying ibis intake, prey biomass and habitat availability (“observation”, “prey-sampled” and “habitat” *trapaengs* respectively). Observation *trapaengs* were also prey-sampled. Inset shows the location of the IBA (dark grey).

spatially and temporally throughout the year (Wright et al. 2010). Drying substrates often crack into polygonal blocks as the dry season progresses, while foraging Wild Boar *Sus scrofa* may also churn up exposed ground (HLW pers. obs.). Although the origin of *trapaengs* is unknown, wild ungulates and domestic livestock may help maintain them: their wallowing removes substrate, and grazing maintains short, sparse vegetation (J.C. Eames and T.D. Evans pers. comm. 2008). Local villagers rely on natural resources in forest and *trapaengs* to sustain their livelihoods (chapter 7); amphibian and swamp eel (Synbranchidae) harvesting for human consumption takes place at *trapaengs* in the dry season.

4.3.2. *Trapaeng* habitat and its availability

To examine the relative extent of *trapaeng* habitat types and their changing availability across the ibis's breeding season, a random sample of 58 *trapaengs* (Fig. 4.2.) were mapped in the early dry season (November 2009) and again in the subsequent mid-dry season (early March 2010); early March coincides with peak chick-provisioning time (Fig. 4.1.). As human activity was expected to have marginal or no impact on habitat extent, this *trapaeng* sample was not stratified by distance to nearest settlement. Within each *trapaeng*, boundaries of homogeneous habitat patches were sketch-mapped with the aid of a hand-held GPS and laser rangefinder (Wright et al. 2010). Habitat patches (861 ± 1501 m^2, mean ± SD) were primarily defined by habitat moisture, including: water (pools or flooded animal wallows); saturated substrate (viscous, liquid mud at pool margins or in wallows); moist substrate (solid but damp earth); and dry substrate (solid with no visible moisture). Within each moisture class, areas with marked differences in vegetation structure (e.g. short, grazed grass versus tall sedge stands) were mapped as separate patches. Moisture class and the height (cm) and cover (%) of vegetation types (comprising grass, sedge, reed, herb and *Sesbania* spp.) were visually estimated for each patch; (as in Arriero et al. 2006; Hill et al. 1990) and one observer (HLW) undertook all mapping and data recording. Maps were georeferenced and digitised in a GIS (ArcGIS 9.3, ESRI 2010) and habitat moisture and vegetation data were aggregated across multiple patches (following Wright et al. 2010) to calculate the percentage extent of habitat types per *trapaeng*. Mean availability of water and substrate moisture conditions at each *trapaeng* was compared between early and mid-dry seasons using paired Wilcoxon tests with Holm adjustment for Type I error rate.

4.3.3. Prey sampling at *trapaengs*

Prey biomass density of exposed *trapaeng* substrates was examined by moisture class and volume of vegetation in habitat patches at 47 *trapaengs* (Fig. 4.2., 55% overlap with habitat-mapped *trapaengs*) in the 2008–09 (n = 20) and 2009–10 (n = 27) breeding seasons. Prey data were collected using soil cores and summed per habitat patch; prey biomass density was not estimated at the *trapaeng* level as placement of cores was not proportional to habitat area within each *trapaeng*. In each year, prey sampling took place evenly across the four-month period late November–early April. *Trapaengs* were randomly selected after stratification by their distance to nearest settlement (0–2.9 km, n = 11; 3–5.9 km, n = 12; 6–8.9 km n = 11; 9–12 km n = 13), as it was anticipated that amphibian and swamp eel harvesting would impact prey biomass (particularly at *trapaengs* close to villages).

However, the survey of *trapaeng* habitat (above) found that only small-scale harvesting (evidenced by substrate levered up from the ground) took place in the study area, occurring at 11 of 58 *trapaengs* (19%) in the mid-dry season and affecting only 4.2 ± 3.3% (mean ± SD) of these *trapaengs*' surfaces; harvesting at prey-sampled *trapaengs* proved even scarcer, and so its effect on prey biomass was most probably negligible.

Prey samples were collected using ten soil cores per *trapaeng*, taking three cores from each substrate moisture class (dry, moist and saturated), or five from each when only two were present. Preliminary observations suggested the ibis rarely fed in water, so waterborne prey and benthic substrates were not sampled. Within moisture classes, cores were taken from a range of vegetation conditions, representative of those observed in *trapaengs* across the study area. Cores measured 25 cm x 25 cm surface area and 18 cm in depth (equivalent to an adult male ibis's bill length), providing 5.29 m^3 total sampled volume across all *trapaengs*. For each soil core, prey type (amphibians, small invertebrates, crabs, swamp eels and snakes), size and count were recorded; amphibian identification followed Neang and Holden (2008). Prey size was classed into body-length intervals of 0–2.49 cm, 2.50–4.99 cm and >= 5 cm; items of < 1 cm were rarely consumed and subsequently excluded from analysis. Means of centigram ash-free dry mass (cg AFDM) per prey type per size class (Piersma et al. 1994), determined from a sample of specimens collected at *trapaengs* (including 21 amphibians, 71 small invertebrates, eight swamp eels and eight crabs), were used to estimate biomass of all prey items.

Mapping of habitat patches and recording of substrate moisture, vegetation cover and height followed the procedures for the wider assessment of *trapaeng* habitat (above). Substrate microtopography (cracked/holed versus even/uncracked ground) was also recorded per soil core. Prey sampling data measure biomass density, but biomass availability to ibis may differ among prey types and substrates, for example being lower in compacted, dry mud that cannot be probed. Although cores were dug rapidly, biomass density may be slightly underestimated as some items could have escaped, e.g. amphibians disappearing into deeply cracked substrates, or swamp eels withdrawing into their burrows within more saturated substrates. The proportionate contribution of each prey type to overall biomass density (cg AFDM totalled across *trapaengs*) was estimated from its contribution in each substrate, multiplied by the average proportionate extent of that substrate type relative to total substrate area at the 58 *trapaengs* mapped in the mid-dry season.

4.3.4. Ibis foraging observations

Ibis activity, habitat use, diet and intake rates were measured at seven of 47 prey-sampled *trapaengs* (Fig. 4.2.) during the 2008–09 and 2009–10 breeding seasons. Observations were spread across a four-month period in each year, from early December to early April, corresponding with the timing of prey sampling. Five *trapaengs* (chosen to cover a range of habitat conditions and to maximise the likelihood of ibis visitation and thus data collection) were sampled in one year and two in both years, providing nine *trapaeng*-year observation periods. *Trapaengs* sampled in both years were observed in different months under different habitat conditions in each year. Observation *trapaengs* were larger with a more even composition of substrate moisture types compared to *trapaengs* across the wider landscape (details in Appendix A, section 4.6.1, Table 4.A1.). *Trapaengs* were situated along a gradient of distance to settlement (range 2.52–9.33 km, 5.62 ± 2.36 km, mean ± SD) and amphibian and swamp eel harvesting had occurred in three of the nine *trapaeng*-years, but once again only very small areas were affected (3.9 ± 3.5 % of the *trapaengs*' surfaces) and anthropogenic prey depletion probably had minimal influence on foraging ibis. Habitat was mapped, recorded and analysed following protocols used across the wider sample of *trapaengs* (above).

One person (HLW) undertook observations from dawn to dusk, for a mean 4.0 ± 0.7 continuous days per site, using a telescope (with 32x magnification) from hides at *trapaeng* perimeters. The observer was typically 2–40 m from foraging birds, allowing prey captures to be seen clearly. Broad spacing of observation *trapaengs* (range 0.85–21.67 km, Fig. 4.2.) improved the likelihood of observing multiple birds, as the ibis disperse widely to breed (HLW unpubl. data). Three *trapaengs*, including the two in closest proximity, regularly held flocks of 10–30 birds, so repeated observation of single individuals could be somewhat minimised; other *trapaengs* typically hosted 1–3 birds.

Intake rates were obtained from replicate six-minute focal samples (totalling 115.2 hours) of adult birds (recently fledged juveniles were excluded). For each capture, the type and size of the prey item was recorded (prey body length visually estimated in relation to ibis bill length) using the categories applied for prey sampling and biomass measurement. Within focal observations, the ibis's use of habitat patches was timed (assisted by markers placed on habitat patch boundaries), enabling habitat-specific intake rates to be calculated. Focal observations rotated or alternated between individuals when more than one bird was present; however, sampling was not fully systematic as it was not possible to track the movements of all birds while obtaining focal data from one individual, and some individuals were observed repeatedly when no other birds were active or present. As none of the habitats used by ibis was densely vegetated, there was no bias towards more easily observed birds or habitats.

Ibis habitat use and activity were recorded by instantaneous scan-samples, scanning all visible individuals at six-minute minimum intervals (typically between focal samples) to record their location (habitat patch identification) and foraging (yes/no). Habitat use was assessed by comparing the proportionate use of habitats (from scan-sample records) to proportionate availability (from digitised habitat maps) across *trapaeng*-years.

4.3.5. Analyses

Ibis dietary composition was estimated from the contribution of each prey type to overall intake rate (across prey types and *trapaeng*-years, AFDM per minute), derived using the proportionate contribution of each prey type to overall intake rate in each substrate type (from focal sample data), multiplied by the proportionate use of that substrate type by ibis, as indicated by scan-samples. Differences in overall ibis intake rate between substrate moisture classes (dry, moist and saturated) were compared with Mann-Whitney tests with Holm adjustment. The effects of substrate moisture and vegetation on ibis intake rates and prey biomass density were then modelled separately for the ibises’ main prey items: amphibians and small invertebrates. As intake rates and prey biomass densities were non-normal and over-dispersed with frequent zeros, both were re-expressed from biomass to count data so that Poisson or negative binomial errors could be fitted. However, prey items were standardised to the equivalent number of prey of the smallest size class (for amphibians, 0–2.49 cm; for small invertebrates, insect larvae of size 0–2.49 cm) using average AFDM, generating biomass-weighted count data.

The effect of vegetation on ibis intake rates and prey biomass density was considered using vegetation volume per habitat patch, a composite index combining all vegetation types into a single metric: $V = \sum (h_i \times c_i)$, where h_i is vegetation height and c_i the proportionate cover of vegetation type i; V was square-rooted in models to reduce leverage. To achieve comparable vegetation volume ranges in ibis intake and prey biomass datasets, ten habitat patches (27 soil cores) of $V > 9$ were removed from the prey biomass dataset. Vegetation volume and substrate moisture variables were confounded (for intake rate model: $r_s = 0.46$, $n = 1927$, $P < 0.001$) with indistinguishable independent effects when included in models together, probably because water scarcity influences vegetation biomass in the dry season.

The effects of these variables were therefore modelled separately, in two rounds: the first compared intake rate and biomass density among classes of substrate moisture; the second investigated the influence of vegetation volume within the moisture class in which ibis intake rate was greatest (dry for intake of amphibians, saturated for intake of small invertebrates).

Intake rate (count of smallest-item equivalents, per habitat per observation, *n* = 1927) and biomass density (count of smallest-item equivalents summed across soil cores per habitat patch, *n* = 159) were modelled in generalised linear mixed models (GLMMs) with Laplace approximation (Bolker et al. 2009). Appropriate error distributions available in the glmmADMB package (Skaug et al. 2012) were compared for each model, and the one resulting in lowest model Akaike Information Criterion (AIC) was selected for final modelling (Bolker et al. 2012). First-round models used a log-normal Poisson error – containing an observation-level random effect to model extra-Poisson variation (Maindonald & Braun 2010) – or negative binomial error distributions with log link; second-round models used log-normal Poisson, quasi-Poisson or negative binomial error distributions (see Appendix B, section 4.6.2, Table 4.A2.). Results were checked for consistency by re-modelling with alternative error distributions and, where possible, using another modelling package (lme4; Bates et al. 2011).

Models of ibis intake rate included the log number of minutes per habitat per observation as an offset, and models of prey biomass density included the log number of soil cores per habitat patch as an offset. Sampling date (days since the sampling season began; random effect) was included in all models to account for potential temporal changes over the four-month sampling periods. Year and site (*trapaeng* ID) were included in all models as fixed and random effects respectively, to account for grouping of prey-sampled habitat patches by *trapaeng* (models of biomass density) or repeat visits to observation *trapaengs* (models of intake). Differences between all substrate moisture classes were tested by alternating the order in which classes were included in the models. Fixed effects were assessed by change in model AIC on their removal (Burnham & Anderson 2002), with an increase in AIC of ≥ 2 units indicating strong support.

The potential magnitude of prey depletion was estimated to examine susceptibility of ibis to intraspecific competition and human exploitation of prey. For a conservative estimate, prey biomass density was compared to the predicted prey consumption by one pair of ibis feeding at one *trapaeng* for 69 days – the duration of the nesting period (Fig. 4.1.) – in the mid-dry season. Depletion was estimated for all consumed prey types (prey biomass density, cg AFDM) and for that part of the diet comprising amphibians alone (count of smallest-item equivalents). Three classes of *trapaeng* size (small, medium, large) were considered, defined by the lower, middle and upper quartile areas of surveyed *trapaengs*. Maximum likely foraging duration per day (assuming 11.8 daylight hours) was estimated from the percentage of scan-samples recording foraging by ibis in four time brackets (0530–0859 hr, 0900–1159, 1200–1459 and 1500–1830) – accounting for varying activity patterns and numbers of observed birds with time of day. Time spent travelling to *trapaengs* or visiting nests for chick provisioning could not be gauged; however, these probably occupy only a small proportion of daily activity and produce only slight overestimation of foraging time. Average intake rates per substrate moisture type were scaled up to *trapaeng* level based on average proportionate use of these types. Prey biomass density averages were scaled up using average proportionate extent of dry, moist and saturated substrates per *trapaeng* in early March.

4.4. Results

4.4.1. Habitat change at *trapaengs*

The extent of water and dry substrate at *trapaengs* changed dramatically from the early to the mid-dry seasons (November–early March, Appendix B, Fig. 4.A1.). Mean water cover dropped from 79.7% to 5.6% over the four months (Wilcoxon test $V_{58,58} = 1711$, $P < 0.001$), while mean dry substrate cover increased from 4.3% to 87.3% ($V_{58,58} = 1711$, $P < 0.001$), by far the most abundant substrate type. Moist substrate extent did not differ between early and mid-dry seasons ($V_{58,58} = 662$, $P = 0.468$) but saturated substrate cover decreased significantly, from 7.9% to 2.2% ($V_{58,58} = 1236$, $P < 0.001$).

4.4.2. Foraging activity and habitat use

Observations at *trapaengs* provided 5122 records of White-shouldered Ibis activity and habitat use (including repeat-observation of individuals) from 1477 scan samples (range 69–287 and mean 146.0 ± 80.5 scans per *trapaeng*-year, ± SD). The percentage of records involving foraging individuals was similar across time-of-day time brackets ($F_{3,32} = 0.84$, $P = 0.484$), averaging 80.0 ± 8.6% overall (mean percent of records per *trapaeng* ± SD). Foraging ibis made negligible use of water relative to exposed substrates, with a mean of only 0.2 ± 0.3% of foraging records (per *trapaeng* ± SD) in pools of water at seven *trapaeng*-years containing both aquatic and substrate habitats. Ibis fed in all substrate moisture conditions (Fig. 4.3.): dry (the commonest), moist and saturated (the scarcest). The proportion of foraging records for each moisture type was variable across *trapaeng*-years but, overall, mean proportionate use of substrates was similar to their mean proportionate availability (Fig. 4.3.).

4.4.3. Composition of ibis diet and prey biomass density

Amphibians, mostly *Microhyla* frogs and Paddy Frog *Fejervarya limnocharis* (typically < 5 cm in body length), formed the majority of the ibis's diet at *trapaengs* (Fig. 4.4.a), providing an estimated 80.6% of overall intake. Amphibians were also the most abundant prey type in *trapaeng* substrates (Fig. 4.4.b), contributing 53.8% of estimated prey biomass density (accounting for average extent of substrate types). Small invertebrates accounted for 9.7% of overall intake compared to 20.0% of prey biomass density. No crabs and only one small swamp eel (Synbranchidae) was caught by ibises, despite together accounting for 21.3% of prey biomass density. Unidentified prey items, probably small invertebrates or parts of amphibians, comprised 8.8% of overall ibis intake.

4.4.4. Influence of habitat on ibis intake rate and prey biomass density

Overall ibis intake rate (pooled across *trapaeng*-years and combining all prey types) varied along the gradient of substrate moisture. Overall intake rate in dry substrate was variable

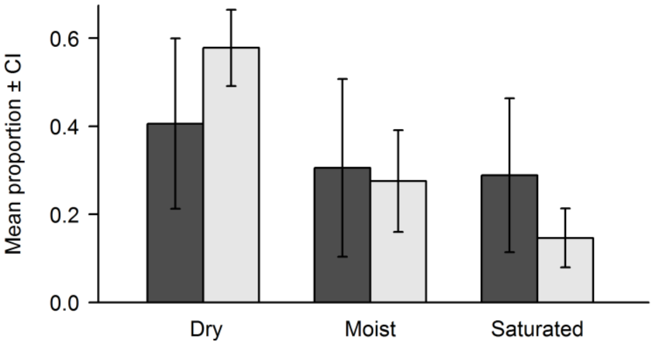

Figure 4.3. Average, proportionate use of substrate moisture conditions by White-shouldered Ibis (dark grey) and their proportionate availability (pale grey) in nine *trapaeng*-year observations. Proportionate use is determined from scan-sampled foraging records and availability is relative to total substrate area per *trapaeng*-year. Bars indicate 95% CIs.

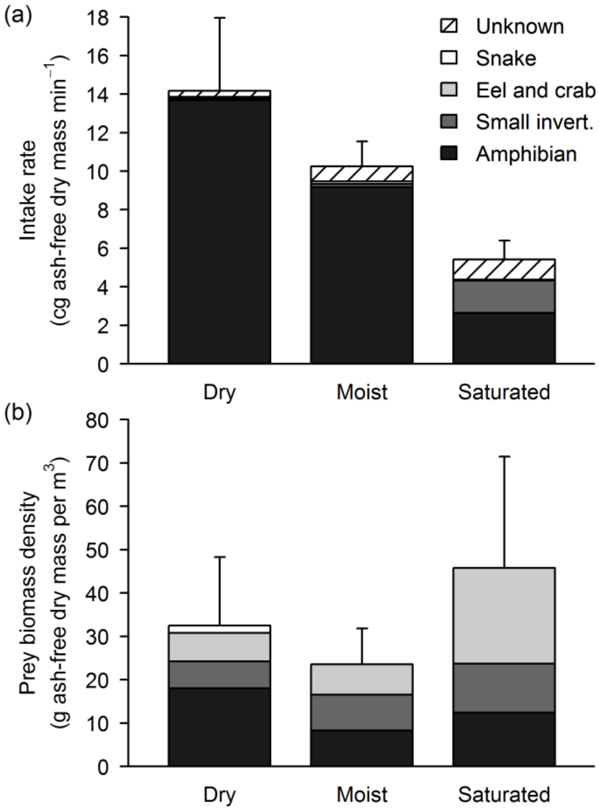

Figure 4.4. Mean White-shouldered Ibis intake rate (a) and mean prey biomass density (b) by substrate moisture types. Mean intake rate is calculated from 676 focal observations in dry, 623 in moist and 628 in saturated substrates (pooled from nine observations at seven *trapaengs*). Mean prey biomass density is calculated from 191 soil cores in dry, 129 in moist and 123 in saturated substrates (pooled from 47 prey-sampled *trapaengs*). Column subdivisions indicate prey type composition as the proportion of overall prey biomass intake rate (a) and overall prey biomass density in each substrate (b); bars indicate 95% CI upper limits. “Small invert.” is small invertebrate; “Eel” is swamp eel.

(Fig. 4.4.a) but marginally higher than that in either moist ($W_{676,623}$ = 197,102.5, $P = 0.063$) or saturated ($W_{676,628}$ = 226,232.5, $P = 0.063$; adjusted for Type I error rate) substrates, while overall intake in moist substrates was greater than in saturated ones ($W_{623,628}$ = 228,497.5, $P < 0.001$; Fig. 4.4.a). Intake rate in saturated substrate was low given that this substrate held a combined biomass density of amphibians and small invertebrates similar to or higher than those in other substrates (Fig. 4.4.b).

Ibis intake rate of amphibians differed between all substrate moisture types, being greater in dry than in moist, and moist than in saturated substrates (Table 4.1., Fig. 4.4.a); these effects were very well supported as model AIC increased by 234 units when the substrate moisture term was removed. Amphibian biomass density was also greater in dry than saturated substrate (Table 4.1., Fig. 4.4.b), and dropping substrate moisture increased this model's AIC by 4.90 units. Within dry substrate, amphibian biomass density was lower in habitat patches with greater square-rooted vegetation volume (Table 4.1., Fig. 4.5.), but vegetation volume showed no effect on intake of amphibians. A supplementary model testing the effect of dry-substrate microtopography found that soil cores in cracked/churned-up dry substrate had greater amphibian biomass density than cores in even/uncracked dry substrate (dropping this term caused a 5.89-unit increase in model AIC; Appendix B, Table 4.A3.).

Intake rate of small invertebrates was greater in saturated (89.5% of intake of small invertebrates) than in both moist and dry substrates (Table 4.1., Fig. 4.4.a), and again the substrate moisture term received strong support, increasing model AIC by 159 units when removed. Small invertebrate biomass density was greater in moist and saturated substrates than in dry substrate and model AIC increased by 8.69 units when substrate moisture was removed from this model. Vegetation volume had no effect on intake rate of small invertebrates or their biomass density in saturated substrate and was not well supported as a model term.

4.4.5. Prey depletion

The scenario of a White-shouldered Ibis pair utilising *trapaengs* over one nesting period (69 days) predicted considerable prey depletion (Table 4.2.). The estimates are conservative as they considered the mid-dry season (early March), when large areas of foraging substrate are exposed. Estimated depletion varied with *trapaeng* size as baseline prey biomass density was proportionate to *trapaeng* area. Depletion rates were greater for amphibians than for all prey combined, a consequence of the ibis's apparent selection for frogs but more limited use of small invertebrates, relative to biomass densities.

(a) Amphibians

Data	Term	Ibis intake rate models		Prey biomass density models	
All	Substrate moisture	ΔAIC = 234.00		ΔAIC = 4.90	
	Dry	–		–	
	Moist	–0.44 ± 0.22[a]	–	–0.45 ± 1.09	–
	Saturated	–1.74 ± 0.22[a]	–1.30 ± 0.26[a]	–1.86 ± 1.41[a]	–1.42 ± 1.51
Dry substrate	Vegetation volume	ΔAIC = –1.00[b]		ΔAIC = 2.00	
		–0.13 ± 0.21		–0.79 ± 0.70[a]	

(b) Small invertebrates

Data	Term	Ibis intake rate models		Prey biomass density models	
All	Substrate moisture	ΔAIC = 159.86		ΔAIC = 8.69	
	Dry	–		–	
	Moist	0.60 ± 0.46[a]	–	0.95 ± 0.59[a]	–
	Saturated	2.59 ± 0.41[a]	1.99 ± 0.42[a]	0.98 ± 0.64[a]	0.03 ± 0.92
Saturated substrate	Vegetation volume	ΔAIC = –1.60[b]		ΔAIC = 1.15	
		0.10 ± 0.31		0.53 ± 0.62	

Table 4.1. Influence of substrate moisture and vegetation volume on White-shouldered Ibis intake rate and biomass density of (a) amphibians and (b) small invertebrates. Substrate moisture was modelled using all data with dry substrate as the reference class (–); comparisons of saturated versus moist substrate (given by re-ordering classes in the model) are shown in separate columns. Vegetation volume effects were modelled in dry substrate for amphibians (676 observations totalling 38.5 hours and 76 habitat patches), and saturated substrate for small invertebrates (623 observations totalling 33.3 hours and 38 habitat patches). ΔAIC is the change in Akaike Information Criterion when the term is dropped from the model. Parameter estimates are given ± 95% CL. Further details are in Appendix B, Table 4.A2. [a] Effects are supported as CI does not cross zero. [b] Negative ΔAIC indicates an improvement in model fit when the term is removed. Positive ΔAIC indicates a deterioration of model fit.

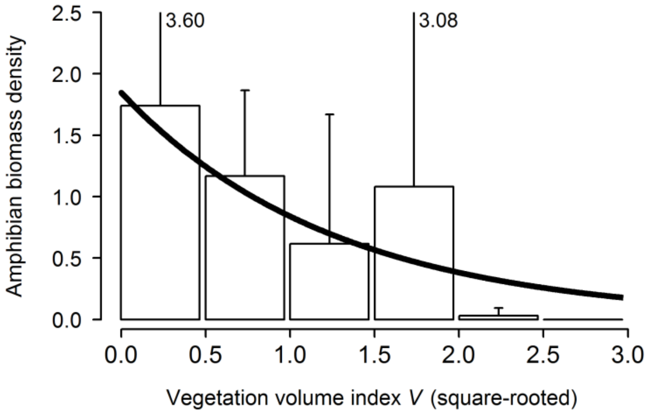

Figure 4.5. Effects of square-rooted vegetation volume on biomass density of amphibians (as a count of smallest-item equivalents) in dry substrate (β –0.79 ± 0.70 95% CL). Solid black line is the fitted relationship predicted by a negative binomial GLMM (based on fixed effects only). Columns indicate mean biomass density for groups of vegetation volume (zero-value column is not missing data). Bars indicate 95% CL upper limits; values beyond the axis range are labelled.

	Prey depletion (%) by *trapaeng* size		
	Small 0.08 ha	Medium 0.18 ha	Large 0.36 ha
All prey types	279.0	126.6	64.01
Amphibians	395.3	179.4	90.7

Table 4.2. Estimated depletion in prey biomass density at *trapaengs* of varying size, using a scenario of *trapaeng* use by a pair of White-shouldered Ibis over one nesting period. Depletion of prey biomass density was considered separately for all prey types (cg AFDM, excluding swamp eels and crabs which were rarely consumed by ibis) and amphibians (count of smallest-item equivalents).

4.5. Discussion

4.5.1. Prey selection and the role of substrate moisture

White-shouldered Ibises strongly avoided aquatic habitats at *trapaengs*, contrasting with the dry season use of water by the sympatric Giant Ibis (Wright et al. 2012a; chapter 5) and the use of diminishing pools by White Ibis *Eudocimus albus* (Russell et al. 2002). Nevertheless, White-shouldered did forage close to water, using substrates exposed by drying pools, mirroring behaviour by Sharp-tailed Ibis *Cercibis oxycerca* and Green Ibis *Mesembrinibis cayennensis* in Venezuelan savannah wetlands (Frederick & Bildstein 1992). As predicted, the ibis foraged in a range of substrate moisture conditions – dry, moist and saturated. Overall, mean proportionate use and availability of these substrates was similar, but their relative use varied among *trapaeng*-years; assessing preference for these habitats is complicated by the close proximity of substrate types within *trapaengs* and the birds' use and movement between them all. Overall intake rate (combining all prey types) did not differ markedly between all substrate types, but intake and prey biomass density of the prey types comprising the majority of ibis diet (amphibians and small invertebrates) revealed stronger effects.

Amphibians contributed eight times more to ibis diet (in terms of biomass consumed) than any other prey type, and formed 26.8% more of estimated overall intake than suggested by their contribution to available prey biomass density (across *trapaengs*, weighted by relative extent of substrate types). This may partly reflect the inaccessibility of swamp eels and crabs, which contributed 21.3% of prey biomass density but were not consumed by the ibis. Swamp eels, for example, are well adapted to drying mud substrates (with fins vestigial or absent) and can escape down burrows (Hill & Watson 2007), probably beyond ibis bill reach, while crabs in saturated substrate may also bury themselves or move to water where ibis do not feed. In contrast, amphibians in firmer, drier substrates may have fewer opportunities to reach safe depths or escape to the ibis's less favourable habitats. When swamp eels and crabs are excluded, amphibians account for 68.3% of available prey biomass density. Modelling intake rate of ibis feeding on amphibians gave strong support for an effect of substrate moisture, with the ibis achieving, on average, 25.8% higher intake of amphibians in dry than in moist substrate, and 78.8% higher intake in dry than in saturated substrate. Although the effect of substrate moisture was supported to a lesser extent in models of biomass density, amphibian biomass was, on average, 31.1% greater in dry than saturated substrate.

These results suggest that dry substrate is of particular importance to foraging ibis. *Trapaengs* may be most attractive in the mid- to late dry season, when this substrate is at least 20 times greater in extent than in the early dry season. Water occupies the majority of *trapaeng* habitat in the early dry season, with the small proportion of exposed *trapaeng* substrates comprising mostly moist and saturated ground. Amphibians may more readily escape from foraging ibis in saturated substrate (e.g. into adjacent pools), perhaps explaining why intake of amphibians appeared disproportionately low relative to their biomass in this habitat. High intake rate and biomass density of amphibians in dry substrate probably relates to the abundance of deep cracks (caused by water drawdown and ground-drying) and holes (created by foraging Wild Boar), where amphibian biomass density was higher than in even/uncracked dry substrate. These cracks and holes may be important refuges for amphibians, providing access to moist, cool conditions away from the hot, dry ground surface.

In contrast to intake of amphibians, intake of small invertebrates was two times less than their biomass density would suggest, contributing only 9.7% of overall intake. Nevertheless, their consumption was not uniform across habitats: models showed strong support for an effect of substrate moisture with more small invertebrates caught in saturated (43.8% on average) than moist (32.2%) and dry (24.0%) substrates. Small invertebrates could be providing a substitute prey source when amphibian resources are depleted and, as substrate penetrability can influence foraging success (Mouritsen & Jensen 1992), ibises may find soft saturated substrate most profitable in finding them (more so than harder, moist substrate where small invertebrates were similarly abundant). However, such a substitution is perhaps independent of seasonal timing, as date was not well supported in additional, exploratory models of intake rate (treating date as a fixed effect). Sexual dimorphism could also be a factor, as the bills of White-shouldered Ibis are, on average, 15% longer in males than in females (N.J. Collar unpubl. data). Females may therefore be less successful at probing for amphibians than for small invertebrates, which are typically caught close to the soil surface (HLW pers. obs.). However, while many ibis species are sexually dimorphic (Babbitt & Frederick 2007), sex-based feeding specialisations appear rare (del Hoyo et al. 1996), and other explanations for use of small invertebrates may also apply, e.g. chicks may require more invertebrates relative to other food sources, as is the case for Black Grouse *Tetrao tetrix* (Baines et al. 1996).

4.5.2. The influence of vegetation on ibis foraging ecology

While a landscape-scale study of habitat use found that ibis preferred *trapaengs* with greater extent of low (< 25 cm) vegetation (Wright et al. 2010), this study found no evidence for a direct effect of vegetation on ibis intake across the range of structures observed. Although it was predicted that tall dense vegetation may restrict ibis habitat accessibility, observation *trapaengs* contained mainly unobstructive low or isolated tussocky vegetation so that such an effect could not be readily tested. Grazing was largely ubiquitous across *trapaengs* within the study area (Wright et al. 2010), making it impossible to observe intake rates of ibis feeding in densely vegetated conditions. A negative effect of vegetation volume on amphibian biomass density was found in dry substrate. This may be caused by the influence of vegetation on substrate cohesion and creation of amphibian refuges; for example, *trapaeng* plant roots are likely to bind soil, limiting the formation of cracks (for amphibians to use) as substrates dry out. Grazing by domestic livestock may influence ibis prey availability where prey biomass is related to vegetation and its removal. Nonetheless, the effect of vegetation observed here appears not to have influenced ibis catch-rate, suggesting that lower amphibian abundance may not correspond with lower availability. Direct and indirect impacts of livestock on ibis foraging ecology deserve further study since well-managed grazing may prove crucial for White-shouldered Ibis conservation (Wright et al. 2010).

4.5.3. A dry-season-adapted strategy

White-shouldered Ibis is clearly adapted to feed in dry season conditions at *trapaengs*. Its avoidance of water but use of all exposed, particularly dry, substrates gives it greater access to food from the mid- to late dry season. The ibis largely feeds on amphibians, the most abundant prey in *trapaeng* substrates, and its curved bill is doubtless an advantage over other waterbirds' short or straight bills for capturing infaunal prey in deep cracks and holes. A curved bill can penetrate further, manoeuvre more easily and withdraw prey intact, as demonstrated for Eurasian Curlew *Numenius arquata* (Davidson et al. 1986; Ferns & Siman 1994). This could explain why White-shouldered Ibis are commonly followed and kleptoparasitised by Cattle Egrets *Bubulcus ibis* and Chinese Pond Herons *Ardeola bacchus* (HLW pers. obs.). The Giant Ibis is the only other species with comparable bill morphology in the White-shouldered Ibis's range, but its wider bill base may make it less effective at exploiting cracked/holed substrate. This may be one reason for its markedly different breeding season (June–November; Clements et al. in press), but study of these species' wet season foraging ecology could provide further explanation.

The White-shouldered Ibis's foraging strategy may explain how it can breed in the peak dry season, an unusual nesting time when most other dry forest waterbirds either do not breed or have already fledged chicks. As exposed substrate appears crucial to *trapaeng* profitability, ibis chick provisioning (peaking in February–March, Fig. 4.1.) coincides with greatly receded water levels. This contrasts with other dry season breeders, such as Lesser Adjutant *Leptoptilos javanicus*, which begin provisioning chicks from November/December (Clements et al. in press). The diets of this and other stork species have a greater fish component than the ibis's (del Hoyo et al. 1996), perhaps necessitating an earlier breeding season when water-levels in *trapaengs* are only partially diminished. Water-levels are known to influence bird breeding strategies, as illustrated by waterbird coloniality and nest timing in the Florida Everglades (Kushlan 1986). Greater Adjutant *L. dubius* may begin provisioning chicks from late December/January (Clements et al. in press), but dry forest populations adopt a different strategy to the White-shouldered Ibis, preferring riverine habitats to *trapaengs* and nesting colonially (T. Clements pers. comm. 2011). Although many ibis species nest colonially (del Hoyo et al. 1996), White-shouldered Ibis breeding pairs are solitary. This may be explained by prey scarcity, forcing the dispersal of pairs across the dry forest landscape to minimise intraspecific competition; breeding adults showed signs of territorial behaviour at foraging sites, with confrontations between birds with nearby nests and other individuals (HLW pers. obs.).

4.5.4. Prey depletion and ibis conservation

The estimated prey depletion by a single pair of breeding White-shouldered Ibis was appreciable, with predicted prey intake exceeding prey biomass density, and amphibian intake nearly double that available, at medium-size *trapaengs*. Prey replenishment could not be considered but is unlikely to be a similar order of magnitude to depletion; *Microhyla* and *Fejervarya* frogs mainly spawn in the wet season (Heyer 1973), making breeding movements and congregations less likely in the late dry season. Although provisional, the estimates of prey depletion suggest that ibis foraging and breeding success is likely to depend on access to multiple *trapaengs*. Accurately quantifying the extent of habitat needed during breeding would require knowledge of the functional response of ibis intake to prey density. However, with a plausible assumption that the drop in intake rate caused by a two-thirds depletion of prey biomass density would force ibis to feed at a different, unexploited *trapaeng*, a breeding pair of ibis may require at least four small (0.08 ha), two medium (0.18 ha) or one large (0.36 ha) *trapaengs* (or a combination thereof), and more if the ibis is largely dependent on amphibian prey.

Prey depletion and scarcity have implications for White-shouldered Ibis conservation. If each breeding pair requires multiple *trapaengs* and intraspecific competition causes population dispersal, habitat needs protection at the landscape scale. Habitat conversion is now the biggest threat to White-shouldered Ibis (BirdLife International 2012b; Wright et al. 2012b; chapter 3), so extensive areas of dry forest must be safeguarded to secure the species's future. Amphibians and swamp-eels are exploited by local people for consumption for most of the year (Allen et al. 2008). While harvesting, at least in the 2009 dry season, proved to be small-scale in the study area, this activity may increase in volume and decrease in selectivity as people respond to declines in other resources, particularly fish (HLW unpubl. data). Such activity will need monitoring as increased harvests, particularly of amphibians, could damage ibis foraging and breeding success. Food resource competition between humans and waterbirds can have disastrous consequences, as the extinction of the Canarian Black Oystercatcher *Haematopus meadewaldoi* has demonstrated (Hockey 1987).

White-shouldered Ibis survival is now closely linked to human activity and its impacts in dry forest landscapes. While the ibis currently benefits from domestic livestock maintaining important foraging habitats (Wright et al. 2010), livelihood change and economic development, bringing different land-use practices and/or more intensive natural resource use (e.g. amphibians), threaten the remaining populations. Even if dry forests and valuable livelihood practices are protected and sustained, climate change may alter rainfall and *trapaeng* hydrology (CEPF 2007), potentially undermining the ibis's dry-season-adapted foraging strategy. Further study of these waterholes, focusing on their formation and optimal configuration, will assist conservation efforts to improve waterhole resilience, benefiting a suite of enigmatic, large waterbirds.

4.6. Appendices

4.6.1. **Appendix A**: Characteristics of foraging observation *trapaengs*

Habitat characteristics and size of *trapaengs* used for ibis foraging observations were compared with a larger, landscape-wide sample (*n* = 58) of habitat-mapped *trapaengs* to assess how well they represented conditions found across the study area (Table 4.A1.).

Some differences between *trapaeng* samples were apparent, with observation *trapaengs* typically larger and having rather more even extent of substrate moisture types (smaller areas of dry but larger areas of moist and saturated substrates) than the landscape-wide sample. The extent of pools of water and the vegetation volume of habitat patches appeared similar between samples. Although observation *trapaengs* do not perfectly represent conditions found across the landscape, the more even composition of substrate moisture types will have aided the assessment of ibis foraging ability across a range of conditions.

	Observation *trapaengs*	Habitat *trapaengs*
Trapaeng area (ha)	1.06 ± 0.68	0.33 ± 0.09
Dry substrate (%)	51.7 ± 11.0	87.3 ± 4.8
Moist substrate (%)	27.1 ± 18.9	4.9 ± 1.8
Saturated substrate (%)	13.7 ± 10.4	2.2 ± 1.0
Pools of water (%)	7.5 ± 7.6	5.6 ± 3.8
Vegetation volume index	4.1 ± 1.7	3.6 ± 4.2

Table 4.A1. Examining potential bias in *trapaeng* size and habitat conditions at ibis foraging-observation *trapaengs*. Mean *trapaeng* area, habitat extents (percentage of *trapaeng* area) and vegetation volume are presented for observation *trapaengs* and a sample of 58 *trapaengs* from across the study area (“habitat *trapaengs*”) surveyed in early March 2010. All nine *trapaeng*-year observations were used to calculate mean *trapaeng* area, but to ensure comparability with data from the habitat *trapaeng* sample, only the five *trapaengs* observed in the late dry season (mid-February to early April) were used to calculate mean habitat characteristics. 95% confidence limits are given as a simple indication of how observation and habitat *trapaeng* samples vary (observation *trapaeng* sample size was too small to conduct reliable statistical tests, although the difference in *trapaeng* area between samples was significant in a Mann-Whitney test, $W_{9,58} = 442$, $P < 0.001$).

4.6.2. **Appendix B**: Additional results and model parameters

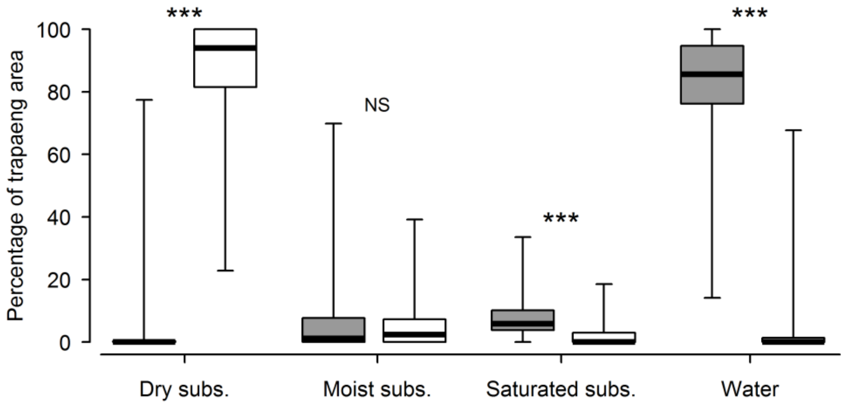

Figure 4.A1. Changes in extent of water and substrate moisture conditions at 58 *trapaengs* between the early (grey) and mid- (white) dry seasons. Asterisks denote significant difference in habitat extent between seasons shown by paired Wilcoxon tests with Holm adjustment ($P < 0.001$); NS indicates a non-significant difference and “subs.” is substrates.

(a) Ibis intake rate of amphibians

All data model. Model AIC = 2879.00 Lognormal-Poisson error with log link				Dry substrate model. Model AIC = 1305.00 Lognormal-Poisson error with log link			
Term	β	*95% CL*	*ΔAIC*	*Term*	β	*95% CL*	*ΔAIC*
Intercept	–1.51	0.82		Intercept	–1.37	1.02	
Moisture			234.00	Vegetation volume	–0.13	0.21	–1.00
Dry	–	–		Year			–1.00
Moist	–0.44	0.22		1	–	–	
Saturated	–1.74	0.22		2	–0.69	0.83	
Year			1.00				
1	–	–					
2	–0.64	0.65					
Random effects: (Date|Site)+(1|ID)				Random effects: (Date|Site)+(1|ID)			

(b) Amphibian biomass density

All data model. Model AIC = 300.40 Lognormal-Poisson error with log link				Dry substrate model. Model AIC = 252.77 Negative binomial with log link			
Term	β	*95% CL*	*ΔAIC*	*Term*	β	*95% CL*	*ΔAIC*
Intercept	–2.67	0.95		Intercept	0.15	0.99	
Moisture			4.10	Vegetation volume	–0.79	0.70	2.00
Dry	–	–		Year			0.45
Moist	–0.35	1.10		1	–	–	
Saturated	–1.77	1.42		2	0.98	1.10	
Year			1.70				
1	–	–					
2	0.93	1.05					
Random effects: (Date|Site)+(1|ID)				Random effects: (Date|Site)			

Table 4.A2. Continued pg. 82.

(c) Ibis intake rate of small invertebrates

All data model. Model AIC = 2896.98 Negative binomial with log link				Saturated substrate model. Model AIC = 2029.00 Negative binomial with variance = θμ, with log link			
Term	*β*	*95% CL*	*ΔAIC*	*Term*	*β*	*95% CL*	*ΔAIC*
Intercept	–4.89	3.55		Intercept	–0.74	0.91	
Moisture			159.86	Vegetation volume	0.10	0.31	–1.60
Dry	–	–		Year			–0.06
Moist	0.60	0.46		1	–	–	
Saturated	2.59	0.41		2	–0.17	0.59	
Year			–1.20				
1	–	–					
2	0.79	2.37					
Random effects: (Date|Site)+(1|ID)				Random effects: (Date|Site)			

(d) Small invertebrate biomass density

All data model. Model AIC = 873.35 Negative binomial with log link				Saturated substrate model. Model AIC = 210.42 Negative binomial with log link			
Term	*β*	*95% CL*	*ΔAIC*	*Term*	*β*	*95% CL*	*ΔAIC*
Intercept	–0.46	0.60		Intercept	–1.37	2.22	
Moisture			8.69	Vegetation volume	0.53	0.62	1.15
Dry	–	–		Year			–4.55
Moist	0.95	0.59		1	–	–	
Saturated	0.98	0.64		2	0.88	1.69	
Year			1.19				
1	–	–					
2	–0.58	0.64					
Random effects: (Date|Site)+(1|ID)				Random effects: (Date|Site)			

Table 4.A2. Parameters of White-shouldered Ibis intake rate and prey biomass density models, for amphibians (a–b) and small invertebrates (c–d) separately. Two models were run for each of a–d, testing the effect of substrate moisture (using all data) and the effect of vegetation volume (within dry substrate for amphibian intake and biomass density, and saturated substrate for small invertebrate intake and biomass density). Error distributions were selected from preliminary tests of model fit with Poisson; lognormal Poisson; zero-inflated lognormal Poisson; negative binomial, zero-inflated negative binomial and negative binomial with variance = θμ (overdispersion parameter x mean, also known as quasi-Poisson) error distributions. Dry substrate and year 1 were reference levels; parameter estimates β are given with 95% CL; ΔAIC is the change in Akaike Information Criterion when the term is dropped from the model. “Date” is the number of days since the sampling season began. The structure of random effects in the model is also indicated, “ID” is the identification number of each observation, creating an observation-level random effect (lognormal Poisson error).

Term	β	95% CL	ΔAIC
Intercept	−2.00	1.65	
Microtopography			5.89
Even/uncracked	–	–	
Cracked/holed	1.83	1.21	
Year			−1.64
1	–	–	
2	0.42	1.37	

Table 4.A3. Parameters of a supplementary model testing the effect of substrate microtopography on amphibian biomass density in dry substrate. The model used a negative binomial error distribution with log link, including date and site as random effects. Even/uncracked substrates and year one were reference levels and parameter estimates β are given with 95% CL. ΔAIC is the change in Akaike Information Criterion when the term is dropped from the model.

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Chapter 5

Foraging ecology of sympatric White-shouldered Ibis *Pseudibis davisoni* and Giant Ibis *Thaumatibis gigantea* in northern Cambodia

Giant Ibises *Thaumatibis gigantea* feeding in shallow water at a *trapaeng*.

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5.1. Abstract

White-shouldered Ibis *Pseudibis davisoni* and Giant Ibis *Thaumatibis gigantea* are two of the most threatened yet poorly known birds of South-East Asia’s dry forests. Anecdotal evidence suggests these species have an intriguing combination of ecological similarities and differences, and as they occur sympatrically there may be an opportunity to safeguard them through joint conservation measures. This study compared their foraging ecology and proximity to people to unravel their ecological differences and inform conservation. Landscape-scale habitat use was assessed by recording ibis sightings on journeys through a 75,000 ha dry forest landscape; White-shouldered Ibises were surveyed over 526 journeys (totalling 17,032 km) and Giant Ibises over 349 journeys (11,402 km). The ibises showed broadly similar habitat selection, using a range of wetland and terrestrial habitats. Giant Ibises were more often sighted further from settlements than White-shouldered Ibises, with maximum sighting frequency predicted at 9.9 km from villages for the former and 8.3 km for the latter. Giant Ibis may be less tolerant of human disturbance and/or White-shouldered Ibis may be more dependent on traditional land management practices, but the species’ differing use of abandoned paddy field (a habitat typically near settlement) could also be a contributing factor. At seasonal waterholes in the dry season foraging Giant Ibis used wetter microhabitats than White-shouldered Ibis suggesting the species occupy different foraging niches. We make preliminary observations regarding Giant Ibis breeding strategy and discuss potential habitat management actions, concluding that although conservation could address these species simultaneously in deciduous dipterocarp forest landscapes, their ecological differences must also be taken into account.

5.2. Introduction

Deciduous dipterocarp forests (“dry forests”) of South-East Asia contain a distinctive assemblage of species including megafauna, such as Asian Elephant *Elephas maximus,* Tiger *Panthera tigris* and Banteng *Bos javanicus*, and large-bodied birds, such as three vulture, four stork and one crane species (Baltzer et al. 2001). This biodiversity has suffered various human impacts, namely hunting, habitat loss and degradation, with at least 60 dry forest birds, mammals and reptiles classified as threatened on the IUCN *Red List* in Cambodia alone (Tordoff et al. 2005; WCS 2009). While conservation resources are being increasingly directed at this ecosystem, the ecology of the forest and much of its wildlife remains poorly or only partially understood (CEPF 2007; Songer 2006). Of the dry forest birds, two species stand out as amongst the most enigmatic, threatened and poorly studied:

the Critically Endangered White-shouldered Ibis *Pseudibis davisoni* and Giant Ibis *Thaumatibis gigantea*.

These two dry forest ibises experienced dramatic declines in the twentieth century (BirdLife International 2001) and, although once widely distributed across South-East Asia, their ranges contracted to become almost entirely confined to Cambodia (BirdLife International 2001). Remaining populations are fragmented and only 250 individual Giant Ibises (BirdLife International 2012b), and 731–856 individual White-shouldered Ibises (Wright et al. 2012a; chapter 3) are estimated to remain globally. Conversion of dry forests (for infrastructure, settlement and agriculture, including plantations) and changing local land management are projected to cause further, severe declines in ibis populations (BirdLife International 2012a; b). Conservation action is urgently required to secure these ibises from extinction, but is likely to depend on a scientific understanding of their ecological requirements.

White-shouldered Ibis and Giant Ibis exhibit an intriguing mixture of ecological similarities and differences. The species occur sympatrically in much of their current ranges (historically they occurred together, or in close proximity, in Cambodia and southern Laos; BirdLife International 2001) and while their wet season foraging ecology remains poorly known (BirdLife International 2012a; Keo 2008b), both forage at seasonal wetlands, known as *trapaengs*, in the dry season (November–May [1], Keo 2008b; Wright et al. 2010). Both ibises breed solitarily in canopies of dipterocarp trees and no evidence of migration has been found for either species. Despite these similarities these ibises have contrasting breeding strategies, with White-shouldered Ibis nesting in the mid- to late dry season (December–May; chapters 7 and 8) and Giant Ibis in the wet to early dry season (June–November; Clements et al. in press). Available evidence suggests that breeding White-shouldered Ibises forage in exposed substrates at drying-out *trapaengs* (Wright et al. 2010). However, why the Giant Ibis breeding season differs so markedly is not yet known, and the habitat use and prey selection of these species has yet to be compared.

The ibises' overlapping ranges and ecology suggest that carefully designed conservation measures could attempt to safeguard both species simultaneously. Conservation that

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[1] Elsewhere in the thesis the dry season is stated as November–April and the wet season May–October. To match the breeding/non-breeding seasons of both White-shouldered Ibis and Giant Ibis, this chapter considers the dry season to be a month longer. In reality there is not a precise or predictable division between seasons and April/May and October/November are transitional periods.

supports local land management practices may benefit both species, particularly the maintenance of foraging habitat by domestic livestock (Keo 2008b; Wright et al. 2010; chapter 6). However, adopting such a strategy requires research into the compatibility of the ibises' ecological requirements and their interaction with people. This study compares White-shouldered Ibis and Giant Ibis foraging ecology, examining habitat selection in the dry forest landscape and microhabitat and prey use at *trapaengs*.

5.3. Methods

5.3.1. Study area

The study was conducted in a c.75,000 ha area within Western Siem Pang and Sekong Important Bird Areas (IBA; centred on 14°17'N 106°27'E), northern Cambodia (Fig. 5.1.,

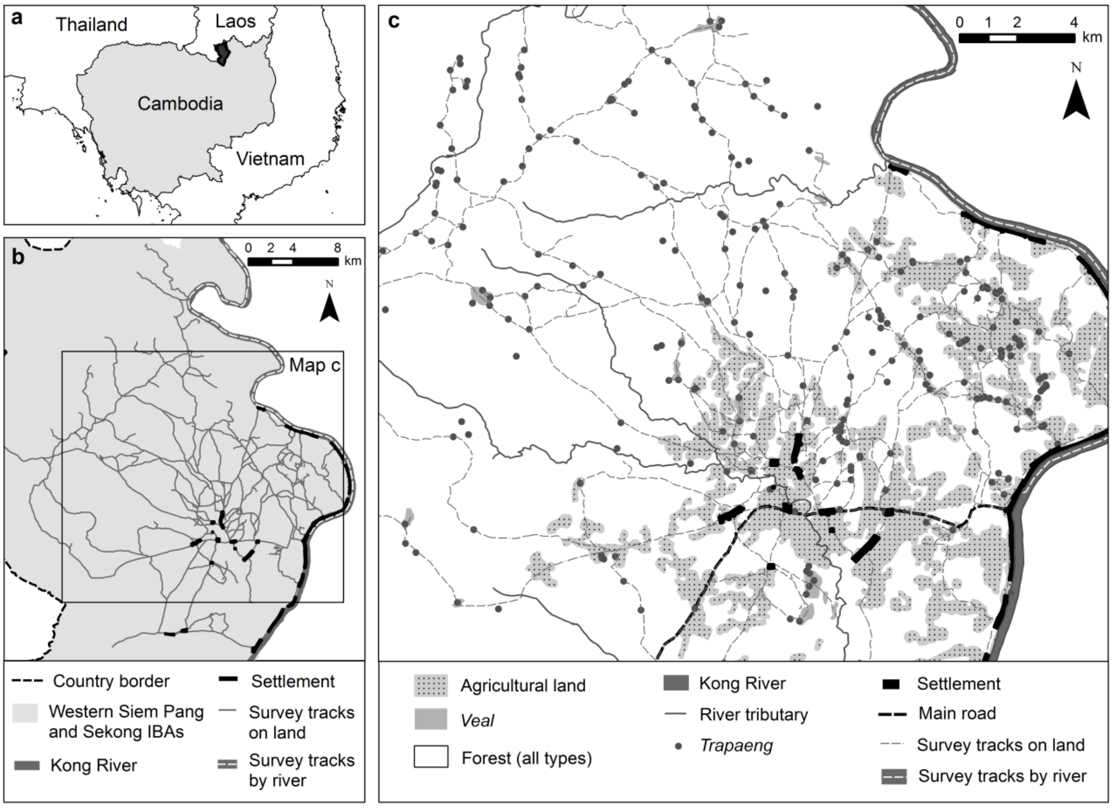

Figure 5.1. Location of Western Siem Pang and Sekong IBAs in Stung Treng Province, Cambodia (a) and extent of survey journeys within the IBAs (b). Map c demonstrates the extent of main habitat types within the core section of the study area. "Agricultural land" includes cultivated/stubble and abandoned paddy fields.

Seng et al. 2003), an unprotected site with at least 226 individual White-shouldered Ibises and an estimated 80 Giant Ibises (BirdLife International 2012b, Wright et al. 2012a; [chapter 3]). The area comprised a mosaic of deciduous dipterocarp forest with patches of grassland (*veals*), river channels, mixed deciduous and semi-evergreen forest, and active and abandoned agricultural land (rice paddy) close to settlements. The climate is strongly monsoonal with monthly average monthly rainfall reaching 333 mm in the wet season and as low as 0.9 mm in the dry season (Thuon & Chambers 2006). Dipterocarp forest understorey is burnt annually in the dry season, largely as a result of anthropogenic fires. *Trapaengs* occur frequently in the landscape and vary in size (0.001–3.4 ha, Wright et al. 2010). Water drawdown in *trapaengs* and rivers is dramatic in the dry season, exposing substrates with varied moisture conditions. Villages were concentrated in the south and east of the study area (Fig. 5.1.) and inhabited by c.10,000 people (Ministry of Planning 2007).

5.3.2. Surveys of ibis habitat use

Ibis habitat selection was examined at the landscape scale by recording ibis sightings along journeys through the study site. White-shouldered Ibises were recorded during 526 journeys over 22 months between November 2009 and January 2012; the protocol was expanded to simultaneously record Giant Ibises, which were surveyed during 349 journeys over 17 months between March 2010 and January 2012 (a subset of White-shouldered Ibis journeys). Journeys were undertaken systematically as part of travel for wider research and on-site conservation activities, with up to three observers travelling independently per day. Journeys were along forest tracks and paths, covering 33.9 ± 18.9 km per journey day (mean ± SD) and were made by motorbike at low speed or occasionally by foot where tracks were inaccessible; 2.4% of journeys were made by boat along main river channels. Journeys were made in both the dry and wet seasons and survey effort (km per journey day) was similar. Survey routes for each journey were noted on datasheets and recorded using a hand-held GPS.

The survey recorded the location (using a GPS), number, activity and habitat use of ibis with each sighting. Ibises on or taking off from the ground were assumed to be foraging and selected for analysis; other activities (such as loafing or preening) may also take place on the ground but are typically interspersed with foraging bouts and occur in the same habitat (HLW pers. obs.). Habitat was categorised as river channel; *trapaeng*; deciduous dipterocarp forest; *veal*; cultivated rice paddy; rice paddy stubbles; abandoned paddy field (unused for more than one season) and mixed deciduous/semi-evergreen forest. The placement of forest tracks was largely independent of vegetation or topographical features, making journeys representative of habitats with the exception of denser semi-evergreen forest (rarely used by either ibis in mainland South-East Asia; BirdLife International 2012a, b), river tributaries and isolated areas of wet-season inundation. Although not traversed, tracks were frequently beside *trapaengs*, allowing them to be surveyed. Much *veal* habitat originated from historic rice cultivation, but swards are typically taller than at more recently abandoned paddies so these habitats were considered separately; bunds were more apparent in abandoned rice paddies (typically > 10 cm high) than in *veals* (typically < 10 cm or absent), allowing these habitats to be distinguished.

Habitats were mapped with a hand-held GPS during journeys in April 2010. Survey effort per habitat type was quantified in a GIS (ArcMap 9.3, ESRI 2010) by intersecting journey tracks onto the habitat segments they traversed. As tracks went beside *trapaengs*, survey effort for this habitat was quantified by intersecting tracks through buffers surrounding each surveyed *trapaeng*. Buffer size was a factor of *trapaeng* radius and viewable distance (in classes of 0, 20, 40, 60 and 80 m) so that large *trapaengs* visible from far away accounted for greater survey effort than small *trapaengs* visible only from close by.

The survey protocol was kept simple so that local field staff and villagers (with low technical expertise) could collect consistent data; as distances to observed birds were not recorded data could not be analysed by a distance-sampling approach. Nonetheless, a preliminary survey did measure the distance from observers to ibises seen on the ground, showing that ibis detectability varied with habitat ($F_{4,34} = 2.71$, $P = 0.046$, distance square-root transformed). Journey distances per habitat segment were therefore multiplied by a habitat-specific estimated transect strip-width, defined by the average sighting distance (or an approximated distance for habitats in which no ibises were observed), to calculate both survey effort and sighting frequency per km^2. Estimated transect widths may slightly underestimate effective strip-width and therefore overestimate sighting density, but this is unlikely to have caused a directional bias in the findings presented and population densities were not estimated. Survey areas were aggregated by habitat type and half-month time periods for analysis.

5.3.3. Foraging observation at *trapaengs* [1]

Ibis microhabitat and prey use were studied at *trapaengs* by observing foraging ibises between December and February in two dry seasons (2008–09 and 2009–10) for seven *trapaeng*-year observation periods: three *trapaengs* in one year and two *trapaengs* in each of the two years. *Trapaengs* were selected to contain a complete gradient of moisture conditions (from pools of water to exposed dry substrate) and maximise the chance of ibis visitation (confirmed by preliminary surveys) to provide foraging data. *Trapaengs* surveyed in both years were observed in different months under novel habitat conditions. Observations were conducted by one observer using a telescope from hides on *trapaeng* perimeters; observations were from dawn until dusk lasting for 3.9 ± 0.7 contiguous days (mean ± SD per site).

Trapaeng microhabitats were defined by four moisture conditions: pools of water, and saturated, moist and dry exposed substrates. Their extents were mapped at each *trapaeng* by sketching homogeneous habitat patches (Wright et al. 2010), recording coordinates with a hand-held GPS and measuring dimensions with a laser rangefinder. Maps were georeferenced and digitised in a GIS to calculate patch areas and area data were aggregated to calculate microhabitat extent as a proportion of each *trapaeng*.

Ibis microhabitat use was measured by instantaneous scan-sampling at six-minute intervals, recording the activity and location (habitat patch) of all ibises present, the latter aided by markers placed around habitat patch boundaries. Prey type and size class (0–2.49 cm, 2.50–4.99 cm and >= 5 cm) of each item captured was recorded for ibis individuals during replicate six-minute focal watches. Items of <1 cm were consumed infrequently and were therefore excluded from analysis. Prey biomass was estimated using average ash-free dry mass (AFDM) calculated for a set of prey specimens comprising all prey types and size classes (Piersma et al. 1994; chapter 4).

5.3.4. Analysis

Landscape-scale habitat selection was examined by log-ratio analysis (Aebischer et al. 1993), comparing proportionate habitat use (from number of sightings) with proportionate habitat availability (from survey effort area) using half-month period as the unit of replication. Analysis was conducted in Compos Analysis software (Smith 2005) with log-

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[1] Protocols followed those described in chapter 4, but sample sizes differed.

ratios weighted by the square-root of total survey effort area per period. Habitat selection was analysed separately for each ibis species in the dry (White-shouldered Ibis n = 23 half-month periods, Giant Ibis n = 15) and wet (White-shouldered Ibis n = 17, Giant Ibis n = 16) seasons. Within the wet season, relative use of *trapaengs* was compared between ibis species with a chi-squared test, contrasting numbers of sightings at *trapaengs* versus non-*trapaeng* habitats by pooling records from journeys made after March 2010 (when both ibis species were surveyed).

To examine the effect of proximity to people on ibis occurrence, ibis sightings and journey tracks were split using a GIS into five classes of distance to nearest settlement (0–2.49, 2.5–4.99, 5–7.49, 7.5–9.99 and 10–16 km). The effect of distance to settlement (midpoints of the five distance classes, treated as a continuous variable) on ibis sighting frequency (count per distance to settlement class per journey day) was modelled for each ibis species in GLMs with Poisson-distributed error and log link, with log survey effort area included as an offset. Sample units with a journey distance of less than 2 km were excluded to ensure counts were based on adequate survey effort. Non-linear effects of distance to settlement were tested by square-root transformation. Proximity of individual sightings to the nearest settlement (calculated in GIS) was also compared between the two species using a Mann-Whitney test.

The species' microhabitat and prey use at *trapaengs* were statistically compared for the *trapaeng*-year observations in which both ibis species were observed, ensuring comparable survey effort and habitat and prey availability. Species' use of dry versus saturated substrate, and of water versus other microhabitats combined, were compared using chi-squared tests of the frequency of scan-sampled individuals per microhabitat type. Intake rate (centigrams of AFDM per minute) of two prey groups: (1) amphibians and small invertebrates; and (2) swamp eels and crabs; were calculated using focal watch data and compared between the two species using Mann-Whitney tests.

5.4. Results

5.4.1. Ibis sighting frequency and flock size

A total of 446 White-shouldered Ibis and 66 Giant Ibis sightings were obtained from 17,032 km and 11,402 km of survey journeys respectively; 328 White-shouldered Ibis and 51 Giant Ibis sightings were of birds seen on or taking off from the ground and assumed to be foraging (“sightings” refers to foraging birds henceforth). Sighting frequency of foraging White-shouldered Ibis per journey day was 0.20 ± 0.40 (mean number of sightings per km^2 ± SD) compared with 0.06 ± 0.31 for Giant Ibis, and mean flock size per sighting was larger (Mann-Whitney test $W_{328,51} = 10{,}142$, $P = 0.011$) for White-shouldered Ibis (5.9 ± 15.3 birds, mean ± SD) than for Giant Ibis (1.8 ± 0.8 birds). White-shouldered Ibis flock size was greater ($W_{99,229} = 5939$, $P < 0.001$) in the wet (non-breeding) season (10.1 ± 25.1 birds) than in the dry (breeding) season (4.1 ± 7.3); no such difference was found for Giant Ibis.

5.4.2. Landscape-scale habitat selection

White-shouldered Ibis and Giant Ibis both foraged in a variety of habitats within the dry forest landscape and both showed marked differences in habitat selection between the dry and wet seasons (Fig. 5.2.). Both ibises preferred *trapaengs* in the dry season, particularly White-shouldered Ibis which breeds in this season. Giant Ibis also made use of river channels in the dry season, a habitat not used by White-shouldered Ibis in this study. In the wet season, ibises made more equal use of habitats and were found more frequently in terrestrial areas than in the dry season. Both species made use of *trapaengs*, *veals* and deciduous dipterocarp forest, but White-shouldered Ibises also used abandoned paddy fields (19.4% of wet season sightings), where Giant Ibises were not observed in either season. *Trapaengs* accounted for a greater proportion of Giant Ibis sightings in the wet season than for White-shouldered Ibis (47% and 22% of sightings respectively) and use of *trapaeng* versus non-*trapaeng* habitat was greater for Giant Ibis than for White-shouldered Ibis ($x^2_1 = 4.01$, $P = 0.045$). Sighting frequency of White-shouldered Ibis at *trapaengs* in the wet season was also 79% lower than in the dry season, compared with only a 34% difference for Giant Ibis. Three habitats appeared unimportant for foraging ibis: one or fewer sightings were gained (for either species) in rice paddy stubbles and cultivated paddy despite the large extent of these habitats in parts of the study area; fewer than three sightings came from mixed deciduous/semi-evergreen forest, perhaps reflecting this habitat’s sparse distribution in the study area and/or poor representation by survey journeys.

5.4.3. Effect of distance to settlement

Ibis sighting frequency per journey day was positively related to distance to settlement for both White-shouldered Ibis and Giant Ibis (Fig. 5.3.), with distance to settlement strongly supported in models for both species; removal of the linear term resulted in Akaike

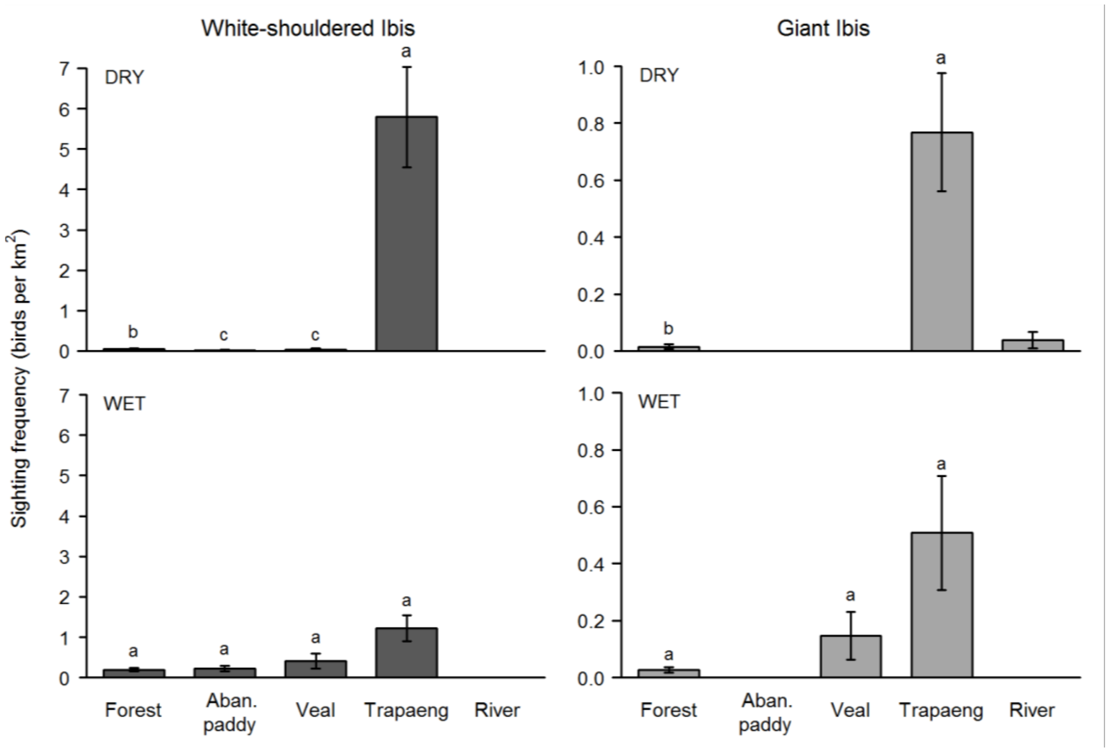

Figure 5.2. Habitat use of foraging White-shouldered Ibis (dark grey) and Giant Ibis (light grey) in the dry and wet seasons. The dry season corresponds with the White-shouldered Ibis breeding season and Giant Ibis non-breeding season, the wet season is the reverse. All habitats were surveyed in each season, missing bars indicate no birds recorded in that habitat type. Sighting frequency is the average number of foraging ibis sightings per km^2 per half-month; bars indicate standard error. Habitat use (log-ratios of use versus availability) did not differ significantly ($P < 0.01$) between habitats sharing a common letter (see Appendix in section 5.6. for mean log-ratio differences and test statistics); river was surveyed too infrequently to analyse its dry-season selection by Giant Ibis. "Forest" refers to deciduous dipterocarp forest, "Aban. paddy" is abandoned paddy field.

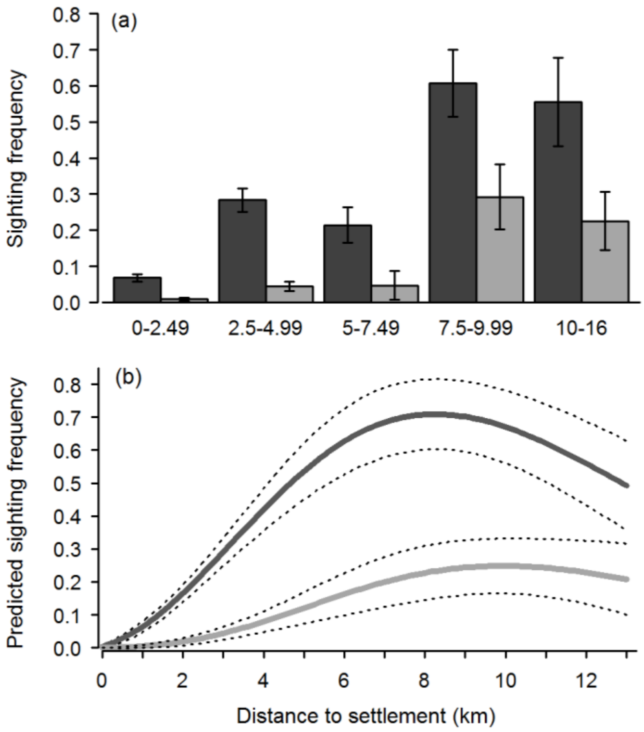

Figure 5.3. White-shouldered Ibis (dark grey) and Giant Ibis (light grey) sighting frequency (number of foraging ibis sightings per km^2 per journey day), averaged by class of distance to settlement (a) and predicted by Poisson GLMs (b): White-shouldered Ibis model AIC = 1585.7, dispersion ratio = 0.77; Giant Ibis model AIC = 376.3, dispersion ratio = 0.31. Bars indicate standard error and dotted lines indicate 95% confidence intervals.

Information Criterion (AIC) increases of > 2 units (63.0 for White-shouldered Ibis and 13.0 for Giant Ibis), removal of the non-linear term resulted in large AIC increases (96.6 and 22.5 respectively). Predicted White-shouldered Ibis sighting frequency had a steeper response curve with distance to settlement (Fig. 5.3.; $n = 1362$, linear term $\beta = -0.68 \pm 0.17$ 95% CL, non-linear term $\beta = 3.93 \pm 0.79$) than Giant Ibis ($n = 904$, $\beta = -0.85 \pm 0.46$, non-linear term $\beta = 5.41 \pm 2.34$); consequently maximum White-shouldered Ibis sighting frequency was predicted at 8.3 km from settlements compared with 9.9 km for Giant Ibis. Further highlighting this difference, the mean distance to nearest settlement of individual White-shouldered Ibis sightings (5.7 ± 3.3, mean ± SD) was significantly less than for Giant Ibis (7.8 ± 3.4, $W_{323,51} = 5474.5$, $P < 0.001$). White-shouldered Ibis sighting frequency appears to decline after 8.3 km from settlements, but lower sample sizes and overlapping confidence intervals (at 8.3 km versus maximum settlement distance) suggest this result is not robust; using additional classes of 10–12.99 and 13–16 km also did not provide strong evidence for a decline (and resulted in even smaller sample sizes).

5.4.4. Microhabitat and prey use at *trapaengs*

White-shouldered Ibis foraging data were obtained in all seven *trapaeng*-year observations and Giant Ibis data in two, yielding 777 and 117 scan-samples (3101 and 242 individual bird records) for these species respectively; scan-samples comprised tens of White-shouldered Ibis individuals and a minimum of eight Giant Ibises. Crude comparison of proportionate microhabitat use shows some apparent similarities between the two ibis species' at *trapaengs* in the dry season (Fig. 5.4.); both fed in all exposed substrate types and showed proportionally greater use of both dry and saturated substrates than moist substrate. However, relative to other microhabitats, Giant Ibis made greater use of pools of water than White-shouldered Ibis ($x^2_1 = 81.6$, $P < 0.001$) for which only 0.3% of scan-sampled individuals (across all *trapaeng*-years) foraged in water. Furthermore, the ibis differed significantly in their use of saturated relative to dry substrate ($x^2_1 = 140.7$, $P < 0.001$), with Giant Ibis making greater use of the former and White-shouldered Ibis greater use of the latter. In terms of proportionate use relative to proportionate availability, White-shouldered Ibis appeared to prefer dry substrate whereas Giant Ibis appeared to avoid it (Fig. 5.4.).

Focal sampling at *trapaengs* yielded 89.7 and 3.46 aggregate hours of foraging observation data (from 797 and 40 focal watches) for White-shouldered Ibis and Giant Ibis respectively. Marked differences in diet composition were apparent, with amphibians contributing greatest biomass for White-shouldered Ibis and swamp eels contributing most to Giant Ibis diet (Fig. 5.5.). At the two *trapaengs* where both ibis species were observed, combined intake rate of amphibians and small invertebrates was significantly greater for White-shouldered Ibis than for Giant Ibis ($W_{349,40} = 3297$, $P < 0.001$). Conversely, combined intake rate of swamp eel and crabs was significantly greater for Giant Ibis than for White-shouldered Ibis ($W_{40,349} = 8532$, $P < 0.001$). Prey biomass estimates for a set of ashed prey specimens suggest the average-sized crab caught by either ibis may hold 2.5 times more AFDM than the average-sized amphibian, and the average-sized swamp eel may hold over 5 times more, indicating that Giant Ibis are likely to be consuming considerably greater prey biomass per item caught than White-shouldered Ibis.

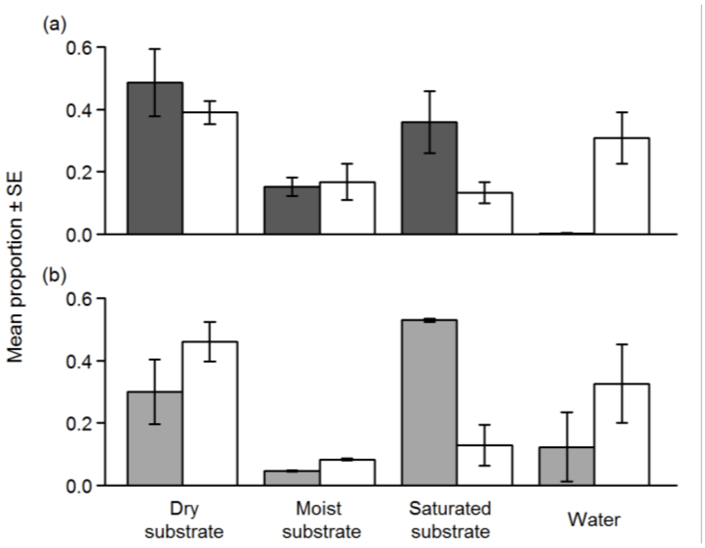

Figure 5.4. Microhabitat use by (a) White-shouldered Ibis and (b) Giant Ibis at *trapaengs* in the dry season. Proportionate use (grey columns) and proportionate availability (white columns) of moisture conditions were averaged across seven *trapaengs* for White-shouldered Ibis and two *trapaengs* for Giant Ibis. Bars indicate standard error.

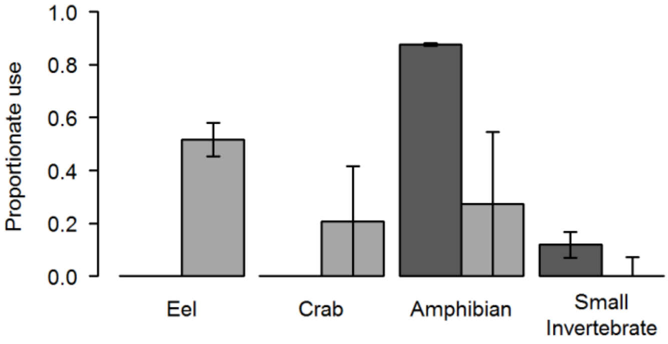

Figure 5.5. Mean proportionate contribution of prey types to biomass consumed by White-shouldered Ibis (dark grey, at seven *trapaengs*) and Giant Ibis (pale grey, at two *trapaengs*) during foraging bouts. Bars indicate standard error.

5.5. Discussion

White-shouldered Ibis and Giant Ibis showed some broad similarities in foraging ecology, including habitat use at the landscape scale. Nevertheless, the relative importance of wetland to terrestrial habitats differed between the species, and contrasting microhabitat and prey use at *trapaengs* suggest these species occupy different foraging niches. Giant Ibises were more often recorded further from settlement than White-shouldered Ibises, suggesting that these species may have different tolerance levels to human disturbance and/or different dependency on traditional land management practices (assumed to predominate closer to settlements) and habitat types. Conservation could be designed to benefit both ibises simultaneously, but will require careful consideration of their ecological differences.

5.5.1. Landscape-scale habitat selection

White-shouldered Ibis and Giant Ibis used a mixture of dry forest habitat types which varied with season. These species showed broadly similar habitat use at the landscape scale in contrast to markedly different breeding strategies. In the dry season, despite rapidly receding water levels (Wright et al. 2010; chapter 4) both ibises preferred to forage in wetland habitats, similar to Sharp-tailed Ibis *Cercibis oxycerca* and Green Ibis *Mesembrinibis cayennensis* habitat selection in the Llanos of Venezuela (Frederick & Bildstein 1992). Mean White-shouldered Ibis sighting frequency was vastly greater at *trapaengs* than in any other habitat in the dry season. This species adopts a foraging strategy well adapted to increasing extents of exposed dry substrates at *trapaengs* in this season (chapter 4) and access to *trapaengs* may be essential for breeding.

Habitat use in the wet season was more equitable, with both ibis species foraging in a range of wetland and terrestrial habitats. *Trapaengs* continued to be used, although to a lesser degree than in the dry season, and the use of open terrestrial habitats (abandoned paddy field and/or *veal*) suggests that access to the ground is important, as for most ibis species reliant on terrestrial habitats (del Hoyo et al. 1996). Keo (2008b) also noted the value of *veals* as a Giant Ibis foraging habitat, observing a high wet-season abundance of earthworms. White-shouldered Ibis was gregarious in the wet season and large congregations of foraging birds (up to 185 individuals) were observed in *veals* and abandoned paddy fields, indicating the importance of protecting such habitats. Apparent habitat preference and response to settlement proximity may be confounded; abandoned paddy fields were not found beyond 4.6 km from settlements, so the absence of Giant Ibis from this habitat may represent avoidance of villages rather than habitat suitability. Conversely the apparently greater tolerance of White-shouldered Ibis for settlement proximity may reflect their greater use of these habitats rather than a differing response to people *per se*.

Although White-shouldered Ibises were not observed in river channels, rivers appear to constitute important foraging habitat elsewhere in Cambodia and Indonesian Borneo (Sözer & van der Heijden 1997; Timmins 2008). Both species are most likely to forage in river channels with large extents of exposed mud and/or sand (R.J. Timmins in litt. 2012); such habitats were rare along surveyed sections of main rivers and seasonal tributary channels in the wider landscape, perhaps containing the most exposed substrate, were poorly represented by journeys along forest tracks. More dedicated survey of suitable riverine habitat would improve knowledge of its selection by both ibis. Nonetheless, the lack of wet-season sightings along rivers may genuinely reflect prohibitively high water levels, an effect seen for White-shouldered Ibis along the Mekong River (Timmins 2008).

5.5.2. Proximity to people

Of the two species, Giant Ibis appeared more limited by proximity to people. Foraging birds were observed significantly further from settlement for this species and in models the predicted maximum sighting frequency occurred 1.6 km further from settlements than White-shouldered Ibis. Keo (2008b) found that Giant Ibis typically nested more than 4 km from settlements and preferred to forage at *trapaengs* further from villages, postulating that disturbance and/or persecution may be greater closer to settlements. White-shouldered Ibis appears much more accustomed to people, often roosting and nesting in trees in or around rice paddies – even when in use by people – and is less wary when approached (HLW pers. obs.). Sightings from the early twentieth century, although sparse, also suggest that White-shouldered Ibis was found more frequently in cultivated lands than Giant Ibis (BirdLife International 2001; Thewlis & Timmins 1996). It is possible that White-shouldered Ibis is more opportunistic in its habitat use, while Giant Ibis may make more specialist use of dry forest landscapes; alternatively these patterns may reflect an underlying difference in responses to human disturbance. Irrespective of the mechanism, Giant Ibis would appear more vulnerable to human activity and settlement in remote areas, while White-shouldered Ibis's closer proximity to people and less evasive behaviour may make it more vulnerable to hunting, a factor that most probably contributed to its decline, particularly in Laos and Vietnam (BirdLife International 2001). Unchecked habitat conversion, resource extraction, human population growth and settlement expansion, issues requiring urgent conservation action in South-East Asia (CEPF 2007), will threaten both species.

Somewhat contrasting dependencies on traditional land management practices could also shape these responses to settlement proximity. While both species could benefit from grazing of foraging habitat by livestock (Keo 2008b; Wright et al. 2010), White-shouldered Ibis is plausibly more constrained by habitat availability, being a third smaller (in terms of linear body length; del Hoyo et al. 1996) and considerably shorter in height than Giant Ibis (HLW pers. obs.). Vegetation growth in *trapaengs*, *veals* and deciduous dipterocarp forest is substantial in the wet season (chapter 6) and may reach more than double White-shouldered Ibis body height (Wright et al. 2010), potentially restricting this species's use of habitats otherwise still accessible to Giant Ibis. More frequent White-shouldered Ibis sightings in areas closer to settlements could reflect a stronger requirement for grazed habitat and bare ground, where livestock densities are highest (Wright et al. 2010). The possible decrease in White-shouldered Ibis sighting frequency beyond 8.3 km from settlements would further support this. The effect of people and livestock on ibis abundance deserves further study to understand the potential trade-off between the negative effects of human disturbance (particularly for Giant Ibis) and the positive effects of land-use practices. Examining ibis distribution in a study landscape with a steeper gradient of livestock density and greater maximum distance to settlement than occurred in this study area would help in clarifying this potential effect.

5.5.3. Foraging ecology at *trapaengs*

The dry season foraging ecology of White-shouldered Ibis and Giant Ibis has been examined elsewhere (Keo 2008a; Wright et al. 2010; chapter 4). Having observed only a small number of Giant Ibises feeding at two *trapaengs*, this study cannot draw major new conclusions on this species's foraging strategy and we limit the discussion to a preliminary comparison of the two species. Similarities included both species' use of all exposed substrate types, and shared use of the amphibian resource at *trapaengs*. Although Keo's (2008b) study (from January to April) found amphibians contributed a major part of Giant Ibis diet (as is the case for White-shouldered Ibis), this study found them to be of only marginal importance. However, our survey took place earlier in the dry season when wetter conditions may have sustained more of the aquatic prey seemingly favoured by this species. The most notable differences in microhabitat use were the avoidance of water and use of dry substrate by White-shouldered Ibis, contrasting with Giant Ibis's greater use of water and saturated substrates. Prey use reflected this with the contribution of aquatic prey to Giant Ibis diet (83% of consumed biomass) far exceeding that in White-shouldered Ibis diet (0.4%). Average biomass of prey items consumed by Giant Ibis was 263% greater than of items consumed by White-shouldered Ibis, perhaps explaining the large disparity in body size between these species, or demonstrating the Giant Ibis's need to consume sizeable prey to sustain its large body size.

Despite some similarities in foraging ecology, the contrasting use of wet microhabitats suggests the ibises may occupy different foraging niches at *trapaengs*, at least in the early to mid-dry season. Studies of ibis incidence at *trapaengs* add further evidence, with Giant Ibis selecting *trapaengs* with greater extent of wet mud (Keo 2008b), but White-shouldered Ibis showing no such selection (Wright et al. 2010). The ibises' morphology may also point to a degree of niche separation: the White-shouldered Ibis's neck is feathered from the body to the top of its hindcrown, whereas Giant Ibis has feathering along only one-third of its neck-length. The latter may be an adaptation for a bird that more regularly submerges its head and upper neck in water or wet mud, although the presence of bare skin may also aid thermoregulation (Buchholz 1996; Ward et al. 2008). When in shallow water, Giant Ibis forage by probing with bill slightly agape (Eames 2011, HLW pers. obs.), matching the technique used by other ibis (e.g. Kushlan 1979) but not witnessed for White-shouldered Ibis.

5.5.4. Giant Ibis breeding strategy

The wet season foraging ecology of these ibises is still very poorly known (BirdLife International 2012a; Keo 2008b); examining the ibises' relative foraging success for wet season prey types and habitats will inform conservation and shed light on the Giant Ibis's breeding strategy. Accessing tall and densely vegetated wet-season habitats may be easier for Giant Ibis (given its larger body size) compared with White-shouldered Ibis allowing it to reach the prey-rich earthworm mounds found in deciduous dipterocarp forest and particularly *veals*. A longer, thicker bill may also be more effective at probing for earthworms. Terrestrial habitats, particularly *veals*, can become inundated in the wet season and *trapaeng* water levels increase substantially (HLW pers. obs.; chapter 4). Greater use of *trapaengs* in the wet season, a greater tendency to forage in wet microhabitats, and longer legs and bill all indicate that Giant Ibis may be better adapted to foraging in flooded conditions. Giant Ibis may therefore have a broader range of suitable wet-season habitat in which to forage and provision chicks.

5.5.5. Implications for habitat management

White-shouldered Ibis and Giant Ibis both used a mosaic of habitat types, requiring protection and management of habitats at the landscape scale. *Trapaengs* are important breeding-season habitats for both species, but a range of terrestrial habitats must also be available, particularly in the wet season. Broadly similar habitat use indicates that conserving a suite of dry forest habitats will benefit these sympatric ibis species simultaneously. The use of open habitats such as *veals* and abandoned paddy fields suggests these features should be protected and kept open; the latter deserves particular attention as it occurs in close proximity to settlement and may be particularly vulnerable to agricultural and urban expansion. Clearings could be created to improve habitat availability in landscapes dominated by dense dry forest, and managed (e.g. by livestock grazing) to ensure they remain accessible. Knowledge of the ibises' wet-season foraging ecology would be valuable to assist the design and management of these open areas.

Maintaining the extensive rearing of domestic livestock is likely to be important to both ibis species – a requirement shared with many other threatened species in pastoral and mixed farming systems (Wright et al. 2012b; chapter 2). Domestic livestock keep sward heights low at *trapaengs*, *veals* and in deciduous dipterocarp forest, maintaining habitat suitability for White-shouldered Ibis (chapter 6). Domestic buffalo are key grazers at *trapaengs* and may also be useful in creating areas of saturated substrate to benefit Giant Ibis (Keo 2008b). Both ibises will be affected if a reduction in livestock causes long-term ecological succession and *trapaeng* sedimentation. Further study should investigate whether introducing buffalo or cattle in landscapes where both domestic livestock and wild herbivores are lacking can improve habitat suitability for these species, and whether animal wallowing may contribute to *trapaeng* creation. Annual dry season fires are another component of traditional, dry forest land management resulting in reduced vegetation height; this could be a benefit to ibises foraging in terrestrial habitats and also deserves further research.

While conservation could benefit White-shouldered Ibis and Giant Ibis simultaneously, care should also be taken to ensure that interventions do not overlook their ecological differences. In particular, safeguarding Giant Ibis is likely to require that large areas of undisturbed habitat are protected from development and human interference, contrasting with the White-shouldered Ibis's potentially stronger need for habitat management. Human disturbance and hunting risk require close attention in interventions that expand, enhance or maintain traditional management practices. Equally, maintaining dry-season water levels at *trapaengs* for the benefit of Giant Ibis or other wildlife (Keo 2008b) should not be undertaken to the extent that the exposed substrates (including dry ground) preferred by breeding White-shouldered Ibis become scarce or unavailable.

5.6. **Appendix**: Habitat selection test statistics

(a) White-shouldered Ibis

Dry season

	Forest	Abandoned paddy	*Veal*
Abandoned paddy	–4.84 ± 1.32 mean ± SE t_{22} = –3.67, P = 0.004	–	–
Veal	–4.43 ± 1.30 mean ± SE t_{22} = –3.41 , P = 0.004	0.41 ± 0.88 mean ± SE t_{22} = 0.47, P = 0.690	–
Trapaeng	7.85 ± 1.73 mean ± SE t_{22} = 4.54, P = 0.002	12.69 ± 1.03 mean ± SE t_{22} = 12.35, P = 0.001	12.28 ± 0.93 mean ± SE t_{22} = 13.30, P = 0.001

Wet season:

	Forest	Abandoned paddy	*Veal*
Abandoned paddy	–2.31 ± 1.83 mean ± SE t_{16} = –1.26, P = 0.225	–	–
Veal	–4.51 ± 1.96 mean ± SE t_{16} = –2.31, P = 0.032	–2.21 ± 2.12 mean ± SE t_{16} = –1.04, P = 0.354	–
Trapaeng	–0.61 ± 1.93 mean ± SE t_{16} = –0.32, P = 0.768	1.70 ± 1.67 mean ± SE t_{16} = 1.01, P = 0.274	3.90 ± 1.95 mean ± SE t_{16} = 2.00, P = 0.087

(b) Giant Ibis

Dry season

	Forest
Trapaeng	8.90 ± 2.06 mean ± SE t_{15} = 4.32, P = 0.003

Wet season

	Forest	*Veal*
Veal	0.26 ± 1.99 mean ± SE t_{14} = 0.13, P = 0.904	–
Trapaeng	2.10 ± 2.52 mean ± SE t_{14} = 0.83, P = 0.428	1.84 ± 1.87 mean ± SE t_{14} = 0.99, P = 0.390

Table 5.A1. Mean differences in log-ratio (of habitat use and habitat availability) between habitat types (rows versus columns), by ibis species (a–b) and season, with *t* test statistic and associated *P* value (calculated using 1000 randomised iterations). “Forest” is deciduous dipterocarp forest.

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Chapter 6

Extensive livestocking practices maintain foraging habitat for a Critically Endangered waterbird

Above: domestic buffalo grazing at a *trapaeng*.
Below: Fire moving through the dry forest understorey.

6.1. Abstract

Developing-world conservation directs much attention towards frontier forest ecosystems where agriculture is considered the greatest threat. Despite the significance of high nature value farming in conservation elsewhere, particularly Europe, less attention is given to traditional farming systems supporting important biodiversity in the developing world. With many of these systems at risk from agricultural modernisation, the traditional practices vital to wildlife require research. The study demonstrates the value of an extensive pastoral farming system to the Critically Endangered White-shouldered Ibis *Pseudibis davisoni*, investigating how grazing and burning provide ibis foraging habitats in a dry forest landscape. Vegetation change was measured following experimental exclusion of livestock at waterholes, and of both livestock and fire in deciduous dipterocarp forest (DDF) understorey; additional field surveys contrasted burnt and unburnt areas of understorey, in the presence of grazing. Grazing and fire effects were related to foraging ibis by analysing the impact of vegetation on ibis incidence, at waterholes and in DDF. Across the study period, vegetation biomass was 92% greater following grazing exclusion at waterholes and 64% greater following grazing and fire exclusion in DDF understorey, where these practices had additive effects. Incidence of foraging ibis was greater in DDF understorey with less vegetation, so that ibis would be 65% less likely to forage in DDF following a > 1.5 year absence of grazing and fire. An impact of grazing on foraging ibis at waterholes was not discernible, most probably because of ubiquitously grazed conditions at waterholes. The study concludes that White-shouldered Ibis is likely to depend on extensive livestocking practices of grazing and burning in Cambodia's dry forests. Conservation must address how valuable practices can be maintained given their imminent, probable transformation in this and other traditional farming systems in the developing-world.

6.2. Introduction

Developing-world agriculture is in transition as the economic forces of globalization respond to growing food demand, increasing affluence and increasing land scarcity (Godfray et al. 2010; Lambin & Meyfroidt 2011). In many places, local, traditional farming systems are being replaced by industrial modes of agriculture as the drive for greater production takes effect (McCullough et al. 2008; Pingali 2007). Large-scale land acquisition, privatisation and the failure to internalise many of agriculture's spillover costs can all lead to major social and environmental problems (Robertson & Pinstrup-Andersen

2010; Weis 2010) including the marginalisation of rural peoples (Cotula et al. 2011; Zoomers 2010) and biodiversity loss (MEA 2005).

Impacts of agriculture have received much attention in conservation science (Norris 2008) and there is considerable debate on reconciling future food demand with biodiversity conservation (Fischer et al. 2011; Green et al. 2005; Phalan et al. 2011). In Europe, agriculture has taken a central place in conservation through the Habitats Directive (advocating the use of low-intensity farming practices to manage semi-natural landscapes) and the widely-applied agri-environmental schemes of the Common Agricultural Policy (Beaufoy 1998; Kleijn & Sutherland 2003). By contrast, conservation in the developing world has given particular focus to forests (Bond & Parr 2010), where agriculture causes substantial biodiversity loss (Sodhi et al. 2010). The dominant paradigm of “countryside biogeography” advocates increasing the permeability and value of the agricultural matrix to benefit forest species (Daily et al. 2001; Ranganathan et al. 2008). However, there is a need to assess how farming can benefit biodiversity in its own right (Adams 2012), and evidence is emerging that agricultural and semi-natural landscapes are vital to a distinct subset of species occurring outside of forests (Wright et al. 2012b; chapter 2).

Where agricultural land uses have mimicked or substituted for the natural habitats they’ve replaced, many open-habitat species, including threatened species of conservation priority, have become dependent on them. Such dependencies are particularly notable in the ancient, traditionally farmed landscapes of Europe, Africa and Asia, but also in recently modified landscapes (Wright et al. 2012b; chapter 2). Farmed landscapes of high conservation value require urgent identification in the developing world, as they are often threatened by both intensification and land abandonment – twin consequences of agricultural modernisation (Donald et al. 2001; Sirami et al. 2008). An understanding of the agricultural practices that underpin synanthropic relationships is needed to inform the design of mechanisms that maintain or mimic valuable land-management systems.

This study considers a tropical deciduous dipterocarp forest (DDF) landscape in Cambodia, an open, savannah-like woodland influenced by livestock grazing and associated dry-season burning in a traditional, extensive farming system. DDF contains some of South-East Asia’s most threatened birds, mammals and reptiles (CEPF 2007), including the Critically Endangered White-shouldered Ibis *Pseudibis davisoni*. Conservation practitioners have postulated that this species may benefit from and perhaps depend on grazing and burning practices, which provide access to its foraging substrates (Timmins 2008; Wright et al. 2010). These practices were experimentally controlled to investigate their impact on wetland and forest understorey vegetation and use these results to explore whether White-shouldered Ibis indeed benefits from extensive livestocking.

6.3. Methods

6.3.1. Study species

White-shouldered Ibis were once widespread in South-East Asia but declined dramatically, for unknown reasons, in the twentieth century; 85–95% of the remaining global population (731–856 birds) occurs in dry forests and along large river channels in north and east Cambodia (Wright et al. 2012a; chapter 3). During the breeding season (December–May) ibis rely heavily on seasonal waterholes, known as *trapaengs*, foraging in exposed substrates around drying pools (Wright et al. 2010; chapters 4 and 5). In the non-breeding season the ibis feed and provision their fledglings in open terrestrial habitats, selecting areas of dry forest with accessible ground (Wright et al. 2010; chapter 5).

6.3.2. Study site

The study took place at Western Siem Pang Important Bird Area (Seng et al. 2003) in northern Cambodia (14°07'N 106°14'E), containing the largest known White-shouldered Ibis population (226 birds; Wright et al. 2012a; chapter 3). Frequent *trapaengs* of 0.001–3.4 ha (Wright et al. 2010) and patches of grassland, mixed deciduous forest and traditional, low-intensity rice field occur within the dominant DDF. A six-month dry season, November–April (with mean monthly rainfall as low as 0.9 mm), is followed by monsoonal rainfall in May–October (up to 333 mm per month; Thuon & Chambers 2006). Approximately 11,000 people live in this 138,000 ha site (Ministry of Planning 2007), depending on livestock rearing, traditional rice cultivation and natural resource harvest (chapter 7). Domestic water buffalo *Bos bubalis* and cattle *Bos taurus indicus* are kept as draught animals and for capital accumulation, and roam freely in the landscape. Cattle typically graze and trample the forest understorey while buffalo graze, trample and wallow (henceforth considered collectively as “grazing”) at *trapaengs* – crucial to both livestock species for water. People manage the forest for livestock by burning most of the understorey in the mid- to late dry season (January–April), encouraging new grass growth after rainfall.

6.3.3. Experimental exclusion of grazing at *trapaengs*

The influence of grazing on *trapaeng* vegetation was examined using 6 m x 6 m timber-fenced exclosures (Fig. 6.1.) over a two-year period. One exclosure was built at each of six *trapaengs* in January–February 2009 (set A) and six more *trapaengs* in January 2010 (set B), creating treatment plots of vegetation left ungrazed for two years and one year respectively. *Trapaengs* were selected to encompass variation in size (0.05–2.32 ha) and vegetation structure (from bare ground to tall dense sedge stands). Few *trapaengs* are burned in the study area so a fire exclusion treatment was not incorporated in this habitat.

Figure 6.1. An exclosure at a waterhole (*trapaeng*) in the early dry season, eight months after construction. Domestic buffalo, and to a lesser extent cattle, have wallowed, trampled and grazed the surrounding *trapaeng* habitat.

At each *trapaeng* the exclosure was paired with a 6 m x 6 m control plot placed within 25 m, in the same habitat, and at a comparable distance from the *trapaeng* centre and margin. Vegetation biomass was monitored monthly, starting in September 2009 for set A control plots and January 2010 for set B. Set A exclosures and paired controls were placed in identical habitat following visual assessment of vegetation structure and type. For set B, where vegetation was measured as fences were built, initial vegetation biomass (square-rooted) did not differ between exclosures and paired control plots ($t_5 = -0.23$, $P = 0.82$).

Vegetation biomass was recorded using a drop-disc (Holmes 1974) weighing 400 g with 25 cm diameter (sufficient to compress rigid Cyperaceae and *Sesbania* stems) and released from the level of uppermost vegetation. Drop-disc height (*dh*, centimetres) provided a surrogate measure of biomass (*b*), as square-rooted dry mass and drop-disc height were strongly related ($b = 0.15dh + 0.09$, $r^2 = 0.56$, $n = 22$, $P < 0.001$) for vegetation collected, dried and weighed at 22 sample points comprising a variety of plant types. To avoid edge effects, drop-disc measures were taken at 21 points on a central 4 m x 4 m grid within exclosures and control plots, excluding the outer metre and four corners of the grid. The predominant vegetation type (grass, rush, sedge, herb, herbaceous bamboo, *Sesbania* spp., shrub and sapling) was recorded at each point unless vegetation occupied < 50% of the drop-disc surface area, when bare substrate was recorded.

Flooded points with submerged vegetation could not be measured by the drop-disc and were given biomass values of half the water depth; in non-flood conditions drop-discs compressed vegetation by 50% on average so half-depth was considered a reasonable substitute. These cases may have led to slight underestimation of vegetation change following grazing exclusion, as mean water depth in the wettest months (September–October) was 3.7 cm higher at controls than in exclosures (Wilcoxon signed rank test $V_{12,12} = 21$, $P = 0.01$). Deeply submerged vegetation may not have been visible but wet-season water was typically shallow (13.9 ± 13.1 cm, mean ± SD of flooded measurement points). Ground rugosity, demonstrated by drop-disc measurements in bare substrate, was significantly greater in exclosures (4.2 ± 3.7 cm, mean of points with < 50% vegetation) than at controls (3.4 ± 2.5 cm, Wilcoxon test $W_{970,1418} = 613769$, $P < 0.001$). The mean difference in rugosity (0.8 cm) was subtracted from all exclosure drop-disc heights to prevent bias; the resulting negative values (0.4% of data points) were set to zero.

To provide comparable vegetation measures to those collected at burnt and unburnt sites across the wider forest landscape and when examining ibis incidence (see below), percentage vegetation cover and average height were visually assessed across the 4 m x 4 m grid in each exclosure and control plot. These data were aggregated into a composite index variable representing vegetation volume: $V = \sum (h_i \times c_i)$ where h_i is vegetation height and c_i the proportionate cover of vegetation type i. Square-rooted vegetation volume V and drop-disc measures of vegetation biomass *dh* were strongly correlated at *trapaengs* (e.g. for January: $r_{24} = 0.78$, $P = 0.01$).

6.3.4. Experimental exclusion of grazing and fire in DDF

The influence of fire and livestock grazing on DDF understorey was examined by comparing vegetation biomass between eight exclosures – where both fire and grazing were prevented – and eight grazed–burnt control plots. DDF sites were selected to represent variable soil condition (sand, gravel and plinthite) and canopy cover (0–25% over exclosures). DDF exclosures were built in January–February 2009 and paired control plots, monitored from September 2009, were placed in similar habitat within 25 m of exclosures; vegetation biomass and vegetation volume were measured following *trapaeng* protocols. Square-rooted drop-disc height was again a strong predictor of square-rooted dry mass ($b = 0.26dh + –0.13$, $r^2 = 0.78$, $n = 22$, $P < 0.001$). Square-rooted vegetation biomass and vegetation volume were strongly correlated (e.g. for August, when the ibis uses the forest, $r_{16} = 0.89$, $P < 0.001$).

Fire was not tested in an independent treatment, because random variation in fire intensity and spread, plus its likely aggregative effects on livestock foraging, cannot be replicated at a small scale: for example, small plots open to livestock but protected from fire may attract disproportionately high grazing levels once surrounded by burnt habitat compared to more typical, larger unburnt patches. Grazing and fire were therefore both simultaneously excluded, with all exclosures protected by 2 m-width firebreaks and all control plots burnt by the dry season fires. Ground rugosity was significantly higher at exclosures (2.8 ± 2.0 cm, mean of points with < 50% vegetation) than controls (2.1 ± 1.5 cm, $W_{110,446} = 19995$, $P = 0.002$); the mean difference (0.7 cm) was subtracted from all exclosure drop-disc heights and resulting negative values (0.4% of data points) were set to zero.

6.3.5. Assessing the effect of fire in DDF

Fire and grazing are complementary management practices in DDF and rarely occur in isolation. Nevertheless, understanding their relative impact on vegetation will inform habitat management for White-shouldered Ibis. To distinguish the effect of fire alone, understorey vegetation was sampled across the DDF landscape in May 2009, at 65 locations burnt by fires and at 34 unburnt locations. Burnt sites were randomly selected in a GIS (ESRI 2006), choosing points within 300 m of forest tracks. Unburnt areas were scarce in

May so they were surveyed systematically whenever new patches were encountered on journeys (within 300 m of tracks) through the landscape. Burning occurs arbitrarily and habitat conditions at unburnt and burnt sites were similar; canopy cover, estimated visually, did not differ between sites ($t_{74} = 0.40$, $P = 0.69$) and neither did cover of permanent earthworm mounds ($W_{65,34} = 886$, $P = 0.15$), a visually estimated proxy of soil condition. Grazing impact, assessed using an ordinal score (0–3) as the proportion of understorey plant stems bitten by animals on a 20 m transect, was also similar between burnt (0.62 ± 0.81, mean ± SD score) and unburnt sites (0.76 ± 0.83; $W_{65,34} = 938.5$, $P = 0.39$). Habitat at each burnt and unburnt site was assessed in four 5 m x 5 m quadrats, placed on alternating sides of the 20 m transect. Percentage cover and average height of vegetation types were assessed visually in each quadrat; trees and saplings over 25 cm diameter at breast height were excluded. Vegetation volume (V, as above) was calculated per quadrat and averaged for each location.

6.3.6. Effect of vegetation volume on ibis incidence

The potential importance of grazing and fire for White-shouldered Ibis foraging ecology was predicted by relating observed vegetation volume differences from the fire and grazing exclusion experiments to models examining the effect of vegetation volume on incidence of foraging ibis (at *trapaengs* and in DDF separately). Data for ibis incidence were obtained in 2008 at 95 *trapaengs* (stratified by size, proximity to settlement and distance to main track) using strip transects to search for distinctive ibis beak marks in the ground (details in Wright et al. 2010). Vegetated habitats were examined with care to avoid bias in beak mark detectability. Ibis foraging locations in DDF were surveyed systematically, using all ibis sightings ($n = 25$) made during journeys in the study site in March–May 2008, and compared to 35 control sites randomly selected (using a GIS) within 300 m of tracks.

Homogeneous habitat patches were sketch-mapped with the aid of a handheld GPS at *trapaengs* and in 20 m x 10 m plots at DDF sites. Percentage cover and average height of vegetation types were visually assessed (trees excluded) to calculate vegetation volume per patch. Mapped patches were subsequently digitised in a GIS to calculate their area (Wright et al. 2010) and vegetation volume was averaged across patches, weighted by their area, to create volume values per *trapaeng* or DDF site.

6.3.7. Analyses

The effect of grazing exclusion on *trapaeng* vegetation and of both grazing and fire exclusion on DDF understorey were modelled in separate generalised linear mixed models (GLMMs) with normal error and identity link, using the lme4 package (Bates et al. 2011) in R (R Development Core Team 2011). Dry season vegetation biomass was modelled for *trapaengs* and wet season biomass for DDF, corresponding to White-shouldered Ibis seasonal habitat preferences (chapter 5). Prior to modelling, exclosure biomass was compared between 2009 and 2010 to test for an inter-annual increase in *trapaeng* vegetation and DDF understorey across the study period; vegetation biomass, averaged per exclosure for the late wet/early dry season (September–January), was compared between years using Wilcoxon signed rank tests (*trapaeng* n = 6, set A exclosures; DDF n = 8). As no difference was found, set A and B exclosure data did not require separate consideration in the model of *trapaeng* vegetation biomass.

Mean vegetation biomass, square-rooted to improve heteroscedasticity, was modelled with treatment, season and year as fixed effects. Treatment at *trapaengs* comprised ungrazed exclosures and grazed controls, and in DDF ungrazed–unburnt exclosures and grazed–burnt controls. Season was divided into time periods that best represented vegetation change while maximising sample size (Table 6.1.). Fixed effects were examined by the change in tested Akaike Information Criteria (AICc) on removal (Burnham & Anderson 2002). Site was included as a random effect in both models; inclusion of site and year controlled for non-independence of data caused by repeated measures across years and the pairing of exclosures and controls at each site (Holt et al. 2011).

The difference in vegetation volume between burnt–grazed and unburnt–grazed DDF sites in the wider landscape was tested using a Mann-Whitney test. To compare the independent effects of fire to the effects of fire and grazing combined, the percentage difference in vegetation volume between burnt–grazed sites and unburnt–grazed sites in May 2009 was contrasted with the percentage difference at burnt–grazed control plots and ungrazed–unburnt exclosures in DDF in May 2010. Comparison of relative rather than absolute differences accounted for uneven rainfall between these years (40% lower in March–early May 2010 than March–early May 2009), but reliable quantification of their importance was impossible.

The effect of vegetation volume (fixed effect, square-rooted to reduce leverage) on ibis incidence was modelled separately for *trapaengs* and DDF using general linear models with

		Fixed effects		Random effect	
Model	Treatment	Season	Year	Site (no. levels) [a]	*n* [b]
Trapaeng	UG vs. G	Late wet (Sep–Nov)	1	6	12
			2	12	24
		Mid-dry (Jan)	1	6	12
			2	12	24
		Late dry (Apr)	1	12	24
DDF	UG, UB vs. G, B	Early wet (May)	2	8	16
		Mid-wet (Aug)	2	8	16
		Early dry (Nov)	1	8	16
			2	8	16

Table 6.1. Structure of models examining the effect of grazing exclusion on *trapaeng* mean vegetation biomass and of both grazing and fire exclusion on DDF understorey vegetation biomass. Site had 12 and eight levels in the *trapaeng* and DDF models respectively, year had two levels in each model. Overall sample sizes: *n* = 94 for *trapaeng* model; *n* = 64 for DDF model. UG = ungrazed, G = grazed, UB = unburnt, B = burnt. [a] Number of site levels per season time period and year. [b] Number of datapoints per season time period and year.

binomial error for presence-absence data. The effects of grazing and fire exclusion on ibis incidence in DDF were predicted by applying mean exclosure and control-plot vegetation volumes to the logistic regression equation of the DDF ibis incidence model. Mean vegetation volumes were calculated using August 2010 data, achieving greatest overlap between the incidence (inter-quartile range 5.90–17.27) and experimental exclusion models (inter-quartile range 6.28–10.85) and representing a time when White-shouldered Ibis forage in DDF (chapter 5).

6.4. Results

6.4.1. Impact of grazing on *trapaeng* vegetation

Ungrazed exclosures had greater vegetation biomass (5.4 ± 8.0 cm, mean difference ± SD, 91.5%) than grazed control plots across the study period (Fig. 6.2.a), and treatment was a strong predictor of *trapaeng* vegetation biomass from late wet to late dry season (Table 6.2.). Vegetation did not accumulate inter-annually, as exclosure biomass was similar in the

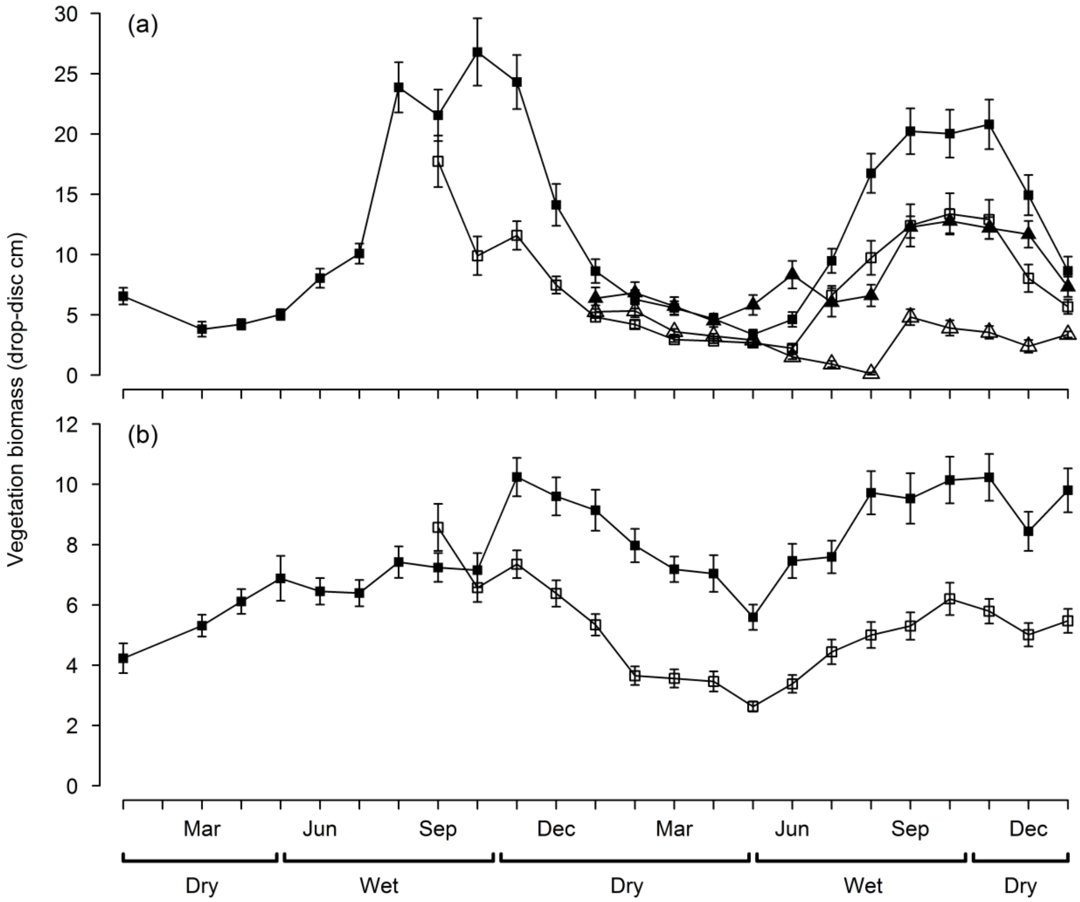

Figure 6.2. Changing mean vegetation biomass following experimental exclusion of grazing at *trapaengs* (a), and grazing and fire in deciduous dipterocarp forest (DDF; b). Six *trapaeng* and six DDF exclosures (solid squares) were built with paired control plots (hollow squares) in 2009 (set A). A further six *trapaeng* exclosures (solid triangles) with paired control plots (hollow triangles) were built in 2010 (set B). Bars indicate standard error intervals; wet and dry seasons are indicated by brackets beneath.

	Trapaengs				DDF		
	β	95% CL	ΔAICc		β	95% CL	ΔAICc
Intercept	3.00	0.64		Intercept	1.61	0.31	
Treatment:			16.94	Treatment:			69.02
grazed	–	–		grazed–burnt	–	–	
ungrazed	0.81	0.34		ungrazed–unburnt	0.73	0.12	
Season:			27.51	Season:			61.53
late wet	–	–		early wet	–	–	
mid-dry	–0.96	0.40		mid-wet	0.66	0.17	
late dry	–1.52	0.55		early dry	0.94	0.17	
Year:			1.69	Year:			0.55
1	–	–		1	–	–	
2	–0.19	0.46		2	–0.15	0.17	

Table 6.2. Parameter estimates for mixed-effects models of square-rooted vegetation biomass at *trapaengs* in the late wet to late dry season (model Akaike Information Criterion [AICc] = 283.10) and deciduous dipterocarp forest (DDF) in the early to late wet season (model AICc = 40.80). Control plots (grazed at *trapaengs*, grazed–burnt in DDF) were the reference level for treatment; late and early wet season were reference levels for season in *trapaeng* and DDF models respectively; year 1 was the reference level for the year term in both models. ΔAICc is the increase in AICc when the term is removed from the model.

late wet/early dry seasons of both 2009 and 2010 ($V_{6,6} = 13$, $P = 0.69$) and year had no effect in the model. However, late dry season rainfall was markedly (40.2%) lower in 2010 than 2009 and, as vegetation biomass probably has a close relationship with rainfall in the late dry season, the possibility of underlying biomass accumulation cannot be discounted. Seasonal differences in vegetation biomass were considerable, with season receiving stronger support than treatment in the model (Table 6.2.). Mean vegetation biomass in exclosures fell by 75.9% between the late wet (October) to late dry (April) season (the result of senescence), while the difference between ungrazed exclosures and grazed control plots was most pronounced in the former time period (10.8 cm mean biomass difference, 119.3% greater) and least apparent in the latter (1.6 cm mean difference, 51.0% greater).

6.4.2. Impact of grazing and burning on DDF understorey

Ungrazed–unburnt exclosures had greater understorey vegetation biomass (3.3 ± 3.2 cm, mean difference ± SD, 63.5%) than grazed–burnt control plots in DDF (Fig. 6.2.b), and treatment was a strongly supported predictor of biomass in the early wet to early dry season (Table 6.2.). Vegetation biomass at exclosures was similar between the 2009 (8.6 ± 3.2 cm) and 2010 late wet/early dry seasons (9.6 ± 4.0 cm, $V_{8,8} = 6$, $P = 0.11$) and year had no effect in the model but, again, between-year rainfall differences may have obscured inter-annual increases in vegetation. Seasonal vegetation change was significant, with mean biomass at exclosures increasing 83.3% from the early wet to early dry season. Mean vegetation volume was 145.0% greater at unburnt–ungrazed exclosures than burnt–grazed controls in the early wet season. This contrasts with the independent effects of fire, which resulted in 34.1% greater mean vegetation volume ($W_{34.65} = 503$, $P < 0.001$, Fig. 6.3.) at unburnt–grazed sites than at burnt, equally as grazed sites across the DDF landscape.

6.4.3. Vegetation volume and ibis incidence at *trapaengs* and in DDF

White-shouldered Ibis incidence was negatively related to vegetation volume at DDF sites (Table 6.3.). The model of ibis incidence in DDF predicted ibis occurrence probabilities (for August) of 0.16 at ungrazed–unburnt exclosures and 0.81 at grazed–burnt controls. In contrast, ibis foraging incidence at *trapaengs* showed no response to vegetation volume across the range observed at unmanipulated sites (Table 6.3.). Without an effect of vegetation volume, ibis incidence could not be assessed relative to the effects of grazing exclusion at *trapaengs*.

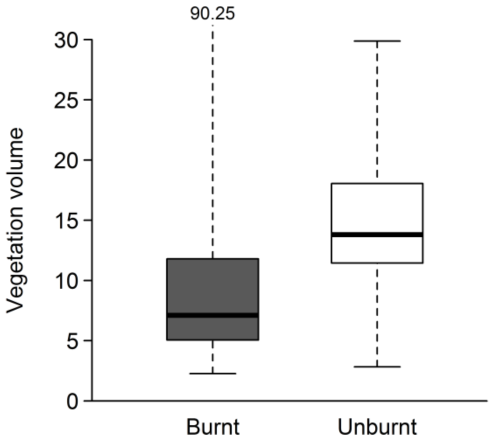

Figure 6.3. Vegetation volume of burnt (grey) and unburnt (white) sites in DDF in the early wet season. Vegetation volume is an index comprising vegetation height and cover. Thick horizontal lines incidicate the median, boxes indicate the interquartile range and error bars (with associated integers) show the extreme values.

	Trapaengs			DDF		
	β	95% CL	ΔAICc	β	95% CL	ΔAICc
Intercept	0.12	0.74		8.75	4.23	
Vegetation volume (square-rooted)	0.17	0.22	0.63	–2.99	1.45	42.66

Table 6.3. Logistic regression model parameter estimates for the influence of vegetation volume on incidence of foraging White-shouldered Ibis at *trapaengs* and in DDF in the late dry season. ΔAICc is the increase in model AICc when the term is removed from the model. *Trapaeng* model AICc = 124.11, dispersion ratio = 1.29. DDF model AICc = 39.61, dispersion ratio = 0.63.

6.5. Discussion

Traditional land management appears to be important in maintaining foraging habitat for White-shouldered Ibis, such that successful conservation of this species is likely to depend on the continuation of extensive livestocking practices. Vegetation biomass was significantly greater with the absence of grazing at *trapaengs*, and with the absence of both grazing and fire in DDF understorey, suggesting these practices help to sustain access to the ground in dry forest habitats. Ibis incidence was negatively related to vegetation volume in forest understorey, and the predicted likelihood of White-shouldered Ibis using DDF to forage was considerably lower with grazing and fire practices excluded. A similar effect at *trapaengs* could not be detected.

6.5.1. Maintenance of short vegetation by grazing and fire

Livestock grazing and anthropogenic fires may shape forest understorey and *trapaeng* habitats in the dry forest landscape. Mean differences across the 17-month exclusionary period showed that *trapaeng* vegetation biomass at ungrazed exclosures was nearly double that at grazed control plots. In DDF, the exclusion of both grazing and fire resulted in nearly two-thirds greater biomass at ungrazed–unburnt exclosures than at grazed–burnt control plots over the same period. Similar effects of fire and grazing have been found by experimental studies in other savannah systems in Asia (Pandey & Singh 1991) and further afield (Hassan et al. 2008; Peco et al. 2006). The apparently greater contrast between exclosures and controls at *trapaengs* than in DDF may relate to greater productivity in the wetland habitat.

Burning of forest understorey and grazing by livestock are co-occurring practices (the former providing fresh, late dry season forage for the latter), but habitat management efforts may benefit from knowledge of their relative, independent importance. The difference in vegetation volume in DDF between ungrazed–unburnt exclosures and grazed–burnt control plots in May 2010 (the effect of fire and grazing combined) was considerably greater than the difference between unburnt and burnt sites in the wider DDF landscape in May 2009 (where fire was assessed in isolation, under equal grazing). This suggests that grazing and fire had additive effects on understorey vegetation, although the between-year difference in rainfall of 40% prevents reliable quantification of their relative importance. High rainfall in 2009 may have caused vigorous vegetation growth at burnt sites, reducing their contrast to unburnt sites in drier years (such as 2010). Additive, and interactive, effects of grazing and fire on sward biomass have been observed elsewhere, such as South African thornveld savannah (Mbatha & Ward 2010); interactive effects are likely in Cambodian dry forests but were beyond the scope of this study.

Inter-annual accumulation of vegetation was not apparent over the study period at *trapaengs* or in DDF, but may have been concealed by rainfall effects. It is unlikely that this study’s exclosures demonstrated maximum potential vegetation growth as they remained young relative to the age of *trapaengs* and DDF stands; a longer study incorporating multi-year rainfall variation may record further increases in vegetation. Nevertheless, the greater biomass observed in the absence of grazing and fire was considerable enough to suggest that short-vegetation habitats would become scarcer without these extensive livestocking practices.

Anthropogenic fires and livestock grazing have long histories in Indochina’s dry forests, with repeated anthropogenic fires, occurring over several millennia, thought to have shaped or encouraged DDF (Maxwell 2004; Stott 1988). Livestock grazing may have gained importance in dry forests more recently, following the twentieth century decline of historically abundant large wild herbivores such as Banteng *Bos javanicus*, Gaur *B. gaurus*, Wild Water Buffalo *B. arnee* and Asian Elephant *Elephas maximus* (Tordoff et al. 2005; Wharton 1968). Domestic livestock may now be mimicking the ecosystem functions that wild herbivores once provided (Timmins 2008; Wright et al. 2010) by grazing, wallowing and trampling at *trapaengs* and in DDF. These actions may also help to sustain key landscape features: wallowing and sediment removal by buffalo may be important to the actual persistence of *trapaengs*.

6.5.2. Role of grazing and burning in ibis foraging ecology

Grazing and burning are likely to have important consequences for White-shouldered Ibis foraging ecology, particularly in DDF where ibis incidence was negatively related to understorey vegetation volume. White-shouldered Ibises were estimated to be two-thirds less likely to feed in DDF in the mid-wet season when grazing and fire have been absent for 20 months. This may represent the ibises’ foraging preference for exposed ground in DDF (Wright et al. 2010), as bare substrate was negatively related to vegetation volume. Habitat accessibility may also contribute, as the ibis stands approximately 35–40 cm high and is likely to be restricted by tall, dense vegetation with gaps narrower than its body width. Improved abundance of prey, such as invertebrates, following fire (Orgeas & Andersen 2001) and grazing (Rambo & Faeth 1999) could also occur and requires research. Similar effects may also occur in dry forest grasslands (*veals*), which contain similar or greater volumes of understorey vegetation than DDF (HLW pers. obs.) and are also used by foraging ibis in the wet season (chapter 5).

The null effect of vegetation volume on ibis incidence at *trapaengs* suggests that the vegetation levels observed in the absence of grazing would not impact foraging ibis. Indeed, ibis were occasionally observed feeding inside some *trapaeng* exclosures (including Fig. 6.1.), despite the significant contrast between exclosure and control-plot vegetation. Ubiquitous, intense grazing at *trapaengs* in the study site, creating sparsely vegetated conditions, may have precluded an effect of vegetation volume on ibis incidence in this habitat. It remains highly plausible that tall dense *trapaeng* vegetation will restrict ibis from foraging on the ground (Wright et al. 2010), and longer studies at sites with lower grazing intensity may reveal the importance of livestock grazing to foraging ibis at *trapaengs*. DDF sites, where livestock densities are probably lower and grazing impact less complete, had a broader range of vegetation conditions than *trapaengs* examined for foraging, perhaps making the effect of vegetation on ibis incidence more readily detectable.

Under present study site conditions, the White-shouldered Ibis is unlikely to be limited by insufficient levels of grazing and burning – most *trapaengs* are open, and dry-season fires burn the majority of the forest understorey. Nevertheless, the importance of extensive livestocking practices may be masked by current conditions, and White-shouldered Ibis survival may depend on them. With few wild herbivores (the natural landscape engineers) remaining, the loss of domestic livestock and reduction of anthropogenic fires would probably cause vegetation to accumulate, inhibiting access to the ground for foraging ibis. A further indication of this relationship may come from White-shouldered Ibis's relative proximity to humans when feeding and breeding, contrasting with other dry forest waterbirds such as Giant Ibis *Thaumatibis gigantea* (Keo 2008). Other threatened birds show similar dependencies on anthropogenic sources of grazing and/or fire, including Bengal Florican *Houbaropsis bengalensis* in Asia (Gray et al. 2007), Liben Lark *Heteromirafra sidamoensis* in eastern Africa (Donald et al. 2010) and Southern Bald Ibis *Geronticus calvus* in southern Africa (Manry 1985). The fate of all these species is closely tied to traditional management that mimics natural ecosystem functions in grassland and savannah-like landscapes.

6.5.3. Maintaining livestocking practices for conservation

The apparent role of extensive livestocking practices in maintaining White-shouldered Ibis foraging habitat illustrates how traditional land management can benefit a threatened species. Encouraging grazing and burning practices in areas of currently unsuitable habitat may assist the ibis's population recovery. Nevertheless, both threatened species and traditional livelihood practices are vulnerable to large-scale land-use change. Conversion of dry forest by external actors to plantation agriculture, infrastructure and settlement threatens both the ibis (Wright et al. 2012a; chapter 3) and traditional forest livelihoods (Baird 2010; McKenney et al. 2004). A win-win strategy, linking the protection of traditional livelihoods to the protection of ibis habitat, may mitigate the threat of major land-use change while sustaining valuable farming practices.

However, even if external threats are alleviated in the short term, socio-economic development may reduce the viability of traditional farming, causing livelihood change and threatening valuable practices in the medium to long term (Wright et al. 2012b; chapter 2). Restoration of large wild herbivores to their former abundance is likely to be a lengthy and difficult process, especially in a context of dry forest habitat loss; therefore valuable ecosystem processes will have to continue being maintained anthropogenically, at least in the short term. Developing-world conservation must urgently design mechanisms that maintain, or mimic, beneficial land management practices in the absence of traditional livelihoods vital in sustaining biodiversity.

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Chapter 7

Linking conservation with livelihoods: a win-win strategy for a threatened waterbird?

Natural resource use is common place in Cambodian dry forests, here a man weaves a basket from strips of wild bamboo and a boy fishes at a *trapaeng*.

7.1. Abstract

A suite of open-habitat species depend on traditional farming practices in the developing world. With no natural habitat remaining, these species are severely threatened by imminent agricultural change. However, there may be opportunities to integrate biodiversity conservation with local livelihoods where rural communities also rely on traditional agriculture and face the same external threats as dependent biodiversity. This is examined for the Critically Endangered White-shouldered Ibis *Pseudibis davisoni*, which benefits from traditional livestock grazing and rice cultivation but is threatened by agricultural land acquisitions in Cambodian dry forests. The study assesses whether livelihoods and the land uses that benefit the ibis are coupled strongly enough to offer a win-win scenario, whereby conservation could uphold local land and livelihood entitlements, benefiting both local communities and the ibis. Income (cash and subsistence) from major livelihood activities and environmental resources was assessed for 64 households for one year; quantitative and qualitative evidence of livelihood change was explored through additional questionnaires, key-informant interviews and focus groups. Average total income was US$333.8 (per person per household) with rice cultivation providing the second greatest income after forest use. Livestock were an important capital asset despite yielding low income; herd value exceeded total income in 48% of households. Environmental income (combining forest use, fishing and livestock rearing) contributed 2.7 times the community's total income from agriculture, demonstrating local dependency on access to the forest. However, while livelihoods benefit from the same farming practices and landscape as the ibis, evidence of livelihood change undermines prospects of a win-win strategy. Agricultural modernisation has accelerated, with a seven-fold increase in hand-tractor purchases in 2005–2010. Patterns of livestock use elsewhere in central Indochina suggest livestock (particularly buffalo) will decline, uncoupling local livelihoods and the ibis. Livelihood transition makes a win-win approach, relying on the persistence of traditional farming techniques, unviable; conservation must therefore seek new mechanisms to maintain or mimic valuable farming practices lost to agricultural change, without compromising local development.

7.2. Introduction

The shared threats of human population growth, ecosystem degradation and climate change to the well-being of people and wildlife provoke continued calls to integrate conservation and development objectives (Kaimowitz & Sheil 2007; Rands et al. 2010; Sachs et al.

2009). The agricultural sector is important to this agenda, occupying 38% of the global land area (World Bank 2012), providing livelihoods for 2.5 billion people in the developing world (FAO 2012), and driving major biodiversity loss (MEA 2005). With predictions of a 70–110% increase in food demand by 2050 (FAO 2009; Tilman et al. 2011), reconciling biodiversity conservation and agricultural development is a major challenge (Norris 2008; Tilman et al. 2002).

Agriculture is often considered simply as a threat to biodiversity in the developing world, and incompatible with conservation (Tscharntke et al. 2005). However, increasing attention is being given to farming systems where human land uses have positive impacts for biodiversity (Fischer et al. 2012; Takeuchi 2010). Evidence is emerging that traditional, low-impact agriculture has become vital to many open-habitat taxa, where farming practices have substituted for lost natural processes (Wright et al. 2012b; chapter 2). With no natural habitat remaining, these species, and the land management they rely on, need urgent conservation in the context of expanding agribusiness and imminent change from traditional to more developed agriculture. Furthermore, this unique biodiversity is dependent on some of the world’s poorest farmers, requiring that conservation also considers the welfare of rural communities, and how they might be incentivised to continue livelihood practices supporting biodiversity (Adams 2012).

Local people may share many of the threats facing farming-dependent biodiversity, particularly agricultural land acquisition by multi-national companies. While some large-scale agricultural investments create livelihood opportunities (e.g. in participatory markets), politically marginalised communities often suffer livelihood change, usurpation of traditional lands and reduced food security (Cotula et al. 2011; Robertson & Pinstrup-Andersen 2010). Where livelihoods rely on the same farming practices and landscapes that support threatened species, conservation could address external threats by defending local land entitlements, mutually benefiting both rural communities and wildlife, at least in the short to medium term (Wright et al. 2012b; chapter 2). Nevertheless, the viability of this “win-win” approach remains untested and uncertain. Theoretical synergisms may oversimplify complex ecological processes and social dynamics in rural landscapes and communities (Brown 2002; Robinson 1993), making win-win conservation strategies difficult to realise in practice (McShane et al. 2011). In contrast to shared extrinsic threats, imminent *intrinsic* change could undermine the win-win scenario. Access to new markets and technology may drive local-scale modernisation of traditional agriculture, making valuable farming practices uneconomic or undesirable to farmers – even if threats from external agribusiness are alleviated. Local agricultural change may therefore uncouple livelihoods and farming-dependent wildlife, putting conservation at odds with human development interests (Wright et al. 2012b; chapter 2).

Research must evaluate the win-win approach by examining the extent to which wildlife and people's livelihoods both rely on the same farming activities and landscapes, and the likelihood of agricultural change that could undermine this linkage. This study undertook this assessment for the Critically Endangered White-shouldered Ibis *Pseudibis davisoni*, a species with only 731–856 birds remaining globally, confined mostly to Cambodia (Wright et al. 2012a; chapter 3). The ibis relies on a variety of habitats modified by extensively-reared livestock and low-intensity rice agriculture, sharing open-access, dry forest landscapes with poor rural communities (e.g. McKenney et al. 2004). Waterholes, a key foraging habitat for breeding ibis (Fig. 7.1.A), were once used by an assemblage of large wild herbivores, including four cattle species and Asian Elephant *Elephas maximus* (Wharton 1968). Following extirpation of these megafauna, grazing and wallowing by domestic livestock now keeps waterhole habitats open, providing access for feeding ibis (Wright et al. 2010; chapter 6). In the non-breeding season the ibis also forages in forest understorey and grassland habitats (Fig. 7.1.B) maintained by livestock grazing and anthropogenic fires (Stott 1986). Abandoned rice paddies, resulting (like many grasslands) from the periodic shift of cultivation to new land, are also used (Fig. 7.1.B). However, dry forest landscapes are now being leased to agribusinesses that seek to replace extensive farming with plantation agriculture, threatening livelihoods (Schneider 2011), compromising poverty alleviation (Sunderlin 2006) and converting globally-important but unprotected ibis habitat (Wright et al. 2012a; chapter 3). Conservation may be able to address this threat to benefit the ibis and local people simultaneously; however intrinsic change may compound conservation problems, as evidence from dry forests in northeast Thailand (e.g. Simaraks et al. 2003) questions the persistence of traditional farming livelihoods.

This paper examines whether the farming activities and dry forest landscape required by White-shouldered Ibis are of core importance to local livelihoods in a community in northern Cambodia. The study quantified the contributions of extensive livestock rearing and rice cultivation to livelihoods, using techniques of household income valuation (Cavendish 2000; Sjaastad et al. 2005) in 64 households for one year to calculate net income (cash and subsistence) for all major livelihood activities. Livestock capital assets

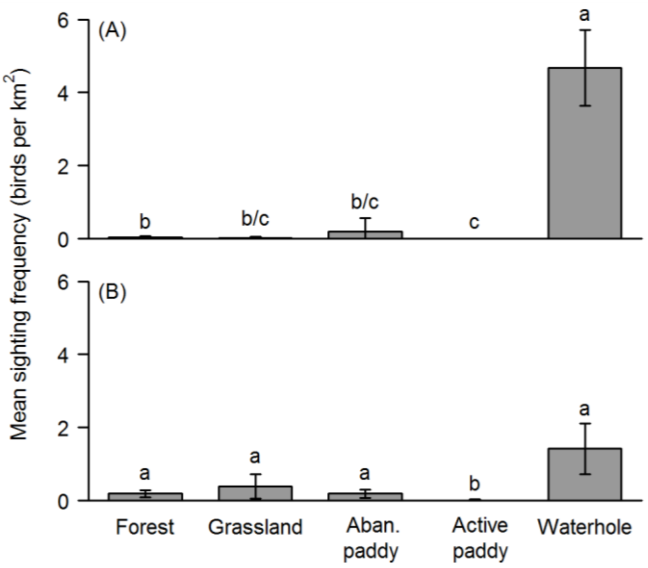

Figure 7.1. Habitat selection by foraging White-shouldered Ibis in the breeding (A, December–May) and non-breeding seasons (B, June–November). Sightings of foraging ibis and their habitat use were recorded on 459 journeys (33.9 ± 16.3 km per journey day, mean ± SD), from November 2009 to March 2011, in a dry forest landscape comprising a mosaic of habitat types. Survey area (km^2) was calculated using transect strip-widths, accounting for differing detectability of ibis with habitat type (see chapter 5 for details). Mean sighting frequency is the number of foraging ibis sightings per km^2 per journey day. Bars indicate 95% CIs. Missing columns indicate extremely low or no sightings. Habitats sharing a common letter did not differ significantly in pairwise Mann-Whitney tests with Bonferroni correction ($P < 0.03$). “Forest” is deciduous dipterocarp forest, “Aban. paddy” is abandoned paddy field.

were quantified to value additional uses of animal herds (e.g. accruing wealth); environmental income (comprising all activities derived from natural resources) was assessed to examine the degree of reliance on the dry forest landscape. The likelihood of livelihood change is investigated by examining agricultural modernisation and the impacts of mechanisation in particular, drawing comparisons with similar Thai farming systems. Together, this evidence was used to assess the viability of a win-win strategy – a useful first examination of conservation mechanisms for high nature value farming systems of the developing world – and highlight the challenges for conserving agriculture-dependent species.

7.3. Methods

7.3.1. Study area

The study took place within Western Siem Pang Important Bird Area (14°07’N, 106°14’E), Siem Pang district, Stung Treng province, Cambodia, an unprotected 138,000 ha area with at least 226 White-shouldered Ibis (Wright et al. 2012a; chapter 3). The site (altitude 55–186 m asl) is dominated by savannah-like deciduous dipterocarp forest, with patches of mixed deciduous and semi-evergreen forest along river channels and at higher elevation. Monsoonal rainfall (1441–2600 mm per annum with 84% falling in the wet season, May–October; Thuon & Chambers 2006) allows cultivation of wet-season rice and flows into the Kong River (a major tributary of the Mekong) running along the site’s eastern edge. Large livestock (buffalo and cattle) roam widely (Wright et al. 2010) relying for much of the year on the forest’s free grazing land.

The study population comprised Siem Pang, a small district town beside the Kong River, and eight other settlements selected systematically in a 9 km radius from Siem Pang , west of the river. Study settlements, containing 7,160 people in total (Ministry of Planning 2007), were in equal proximity to the forest and were all beside the only main road or Kong River. Siem Pang had a small market trading basic goods with surrounding villages. The nearest large market town (Stung Treng, 80 km away) was reached by seasonally-passable forest tracks, or by boat on the Kong River, until construction of a main road in 2009–2010 allowed year-round road transport. Families had no formal land title to the forest or to most of their paddy fields. Use of dry forest resources was largely unregulated although small-scale, poorly-resourced law enforcement by the local Forestry Administration confiscated illegally-logged timber.

7.3.2. Livelihood data collection

Livelihood activities and products (both inputs and outputs) were identified in a scoping questionnaire, undertaken in the dry season (January–April 2009) at one in three households (*n* = 258) in the nine study settlements; adult household members were interviewed and when absent the next nearest household was selected. This scoping questionnaire provided initial contact with the community and contextual information to guide subsequent household income valuation. Six main livelihood activities were identified: forest resource use; fishing; extensive livestock rearing (raising buffalo *Bos bubalis* and cattle *Bos taurus* *indicus* in the open-access forest and requiring little input); garden produce (fruit, vegetables and small-scale maize and sugarcane crops) and animals (monogastrics such as pigs, ducks and chickens); rice cultivation; and wage labour (agricultural, government or non-governmental organisation [NGO]) plus business (small-scale transportation or local market stalls). This list was confirmed through 23 semi-structured key informant interviews with villagers, government officials (e.g. at village, commune and district levels) and local development NGO staff (henceforth all are termed “informants”); these interviews also provided additional contextual information, such as recent livelihood trends and the extent of illegal activity.

Household incomes were subsequently quantified using structured questionnaires at 70 households selected randomly from the scoping questionnaire sample; attrition resulted in a final sample of 64, but with no evidence of bias caused by the loss of six households. Income sources were assessed over one year by two visits to each household, occurring in the late rice cultivation season (November–December 2009) and late dry season (April–May 2010). This provided a snapshot of livelihoods, but informants helped to identify atypical characteristics of the study year. A pilot questionnaire was developed, translated and tested at households outside of the sample. Questionnaires were conducted with a single adult member of the household (allowed to differ between visits) informed about the purpose of the study. Surveys lasted approximately 45 minutes and involved a similar number of men (45.2%) and women (54.8%).

Questionnaire respondents (henceforth “respondents”) were asked to quantify the household’s use of products for six-month recall periods, considering cash/barter income, consumption or use as inputs into other activities. Use of subsistence products, especially those collected opportunistically, may be difficult to recall over long time-periods (Lund et al. 2008; McElwee 2008) making their quantities approximate and perhaps underestimated; however, products of trivial importance (e.g. medicinal plants and remittances) were excluded. Respondent anonymity was assured and while 63% of households provided data on forest products collected illegally (protected animals and high-grade timber), hesitancy by some respondents suggested these products may be underestimated. The questionnaire also collected data on household demographics; livestock herd sizes, gains and losses; and reasons for selling livestock.

All questionnaires were undertaken by a trained Cambodian graduate (SP) with experience of social research techniques; this author spent 1.5 years living in the community, gaining a degree of local trust. A local Lao-speaking translator was also used when necessary.

Nobody of recognised authority was present during data collection, minimising the likelihood that people withheld information through fear of prosecution or disapproval. Responses were checked thoroughly during data entry and households were revisited to clarify inconsistent or evidently erroneous data. Repeat visits to households enabled information to be cross-checked, particularly when respondents differed from previous visits.

7.3.3. Assessing livelihood change

An apparent increase in the use of two-wheeled “hand-tractors” for transport and ploughing was investigated by both quantitative and qualitative techniques, determining the scale and potential impacts of this mechanisation process. In February 2011, a structured questionnaire was administered at one in five households in each study settlement ($n = 150$ in total) to collect data on hand-tractor ownership, purchase date, cost and livestock sold. Recall periods were up to 11 years, but hand-tractor purchases were a major investment and proved very memorable to respondents. Focus groups of tractor-owners were held in three villages (five participants in each), using semi-structured interview guides to discuss the reasons for purchasing hand-tractors and the subsequent uses of livestock. Evidence was triangulated by 21 key informant interviews (a subset of the informant interviews described above), including: a vet; five livestock-owning villagers; two abattoir owners; three hand-tractor traders and mechanics; six government officials; and four local NGO staff. Questionnaires were undertaken by a Cambodian graduate. The lead British researcher (HLW) was present at focus groups and key informant interviews.

7.3.4. Household income accounting and analysis

Household income from each main livelihood activity was calculated over the year using the “value-added” approach of Sjaastad et al. (2005): net income = gross value of outputs minus the value of all inputs and costs. Subsistence use was included using cash-income equivalent values; total net income (henceforth “income”) is therefore a monetary representation of livelihood value (not cash profit) and negative incomes (deficits) occurred when total input value exceeded total output value. Barter transactions (e.g. ploughing draught paid for in rice) were valued using the cash-income equivalent of the payment item. Monetary values were assigned to all products and transactions using either the household’s reported cash income values, or average local farm-gate prices applied to product quantities (Cavendish 2000).

Local farm-gate prices were obtained by: market surveys in Siem Pang (at the beginning and end of the study year); a focus group of six local producers and traders; and qualitative interviews with key informants. Most products had a local farm-gate price (US$1 = 4000 Riel) with the exception of fuelwood, fence poles and rice-straw livestock fodder, which locals considered to be freely available; these were given imputed values based on respondents' willingness to pay for labour to collect them (following Kamanga et al. 2009). Local prices were used to check the validity of cash income data from respondents; local price data and household's own-reported values were broadly consistent and therefore combined to obtain average prices for each product.

Where inputs to one livelihood activity originated from another, the input was deducted from the new activity's gross value and rebooked as an output value of the originating activity (Cavendish 2002; Rayamajhi et al. 2012). For example, the value of households' own livestock ploughing draught was deducted from rice cultivation but added to extensive livestock rearing. A summary of livelihood activity inputs and outputs is given in section 7.6.1 (Appendix A, Table 7.A1.).

The value-added approach does not deduct household own-labour from net income; estimating such costs is difficult when labour markets are minimal (Babulo et al. 2009) but, as skilled labour contributed little to livelihoods in this study, the comparison of income sources was not invalidated. Nevertheless, incomes are presented divided by households' time-weighted adult equivalent units (AEU), a measure of household size calculated by summing members' adult equivalent units – derived from Murthi's (1994) coefficients of adult equivalence per sex and age group – multiplied by the proportion of the study year that they lived in the household (following Cavendish 2002).

Income per household was compared between livelihood activities using a Friedman test (accounting for household repeated-measures) with Bonferroni-adjusted Wilcoxon tests for post-hoc analysis. The household sample was then divided into terciles of total income, allowing comparison of income per livelihood activity between poor (total income below US$166.9 per AEU, n = 21), medium-income (US$166.9–340.2, n = 21) and rich (> US$340.2, n = 22) households using Kruskal-Wallis tests with Bonferroni-adjusted Mann-Whitney tests for post-hoc analysis. Overall income share per livelihood activity was calculated as the percentage of aggregated income (summed across all households in the sample or given tercile); negative income values prevented income share calculations at the household level.

For livestock, capital-asset value (herd capital) was calculated for the start of the study year using herd demography data and average local prices per sex, age and animal type. Pigs and fowl reared at the home were excluded. The asset value accrued over one year (capital change) was calculated from the net result of value gains (births, recruitment and purchases) and losses (deaths and sales), for livestock-owning households only. Herd capital, capital change, births and recruitment, purchases, deaths and sales were each compared between household income terciles by Kruskal-Wallis and Mann-Whitney tests.

7.4. Results

7.4.1. Value of livelihood activities across households

Income over the one-year study period was variable among households (US$333.8 ± 294.2 per AEU per household, mean ± SD). Although most households undertook several (4.7 ± 0.8) livelihood activities, the six main activities (Fig. 7.2.) were undertaken by > 80% of households with the exception of wage labour and business (20.3%). The number of activities undertaken did not differ among income terciles ($F_{2,61}$ = 0.58, P = 0.560), but was positively correlated with household AEU (r_s = 0.29, n = 64, P = 0.020). Household size

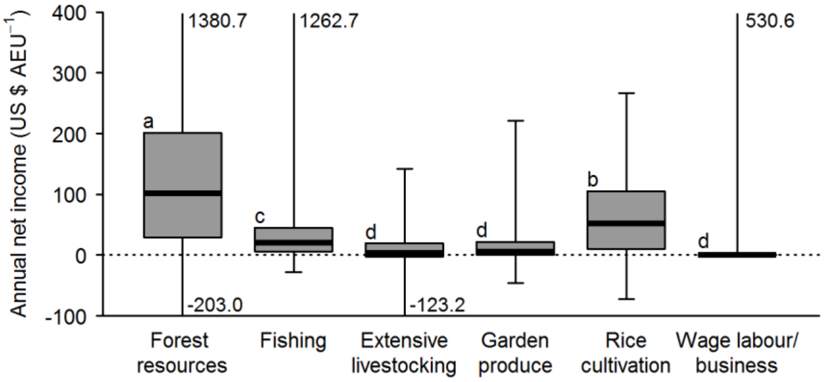

Figure 7.2. Net income by livelihood activity in 64 households for a one-year period (US$ per time-adjusted adult equivalent unit, AEU). Thick horizontal bars indicate the median; minimum/maximum values beyond the axis range are labelled. Boxes sharing a common letter are not significantly different (P < 0.03) in Wilcoxon tests with Bonferroni correction; see section 7.6.2. (Appendix B , Table 7.A2.) for test statistics. Garden produce includes animals reared at home (e.g. chickens, ducks and pigs).

was even across income terciles (5.7 ± 1.9 members overall, $F_{2,61} = 0.63$, $P = 0.53$), as was the mean age of members per household (22.7 ± 7.5 years, $F_{2,61} = 0.14$, $P = 0.873$). Nevertheless, time-adjusted AEU differed ($F_{2,61} = 4.90$, $P = 0.011$), with rich households (AEU 4.3 ± 1.5) having lower standardised labour than poor households (AEU 5.8 ± 1.9), primarily owing to more female members. Rice paddy hectarage was similar between income terciles at 1.7 ± 0.9 ha per household.

Income differed between livelihood activities (Fig. 7.2.; Friedman $x^2_5 = 95.52$, $P < 0.001$), with forest resources providing highest income (48.1% of total income across households), followed by rice cultivation (20.2%) and fishing (15.2%); extensive livestock rearing created much less income (2.4%). Key informant evidence suggested that collection of malva nuts and timber (construction and illegal high-grade wood) from the forest was unusually high in the study year (5/5 stated it was greater than in the previous 2–5 years). With these products excluded, income from routine forest use (e.g. fuelwood, animals, wild foods and other plants) remained high (interquartile range US$22.9–81.3) and significantly greater than for fishing (Wilcoxon $V_{64,64} = 1530$, $P = 0.001$) but not rice cultivation ($V_{64,64} = 930$, $P = 0.464$). Subsistence use accounted for most fishing (95.5 ± 14.0%), livestock (94.1 ± 21.8%) and rice (92.4 ± 13.5%) products; forest and garden products were split between subsistence (47.2 ± 36.8% and 49.4 ± 34.1%, respectively) and cash-income uses. The study community gained US$14,038.1 total environmental income (combining forest resource, fishing and livestock rearing incomes from all sampled households) compared to US$5230.7 from agricultural activities (garden produce and rice cultivation); total environmental income excluding malva nuts and timber was US$8020.3.

Poor, medium-income and rich households gained different incomes from forest resource use (Kruskal-Wallis $x^2_2 = 35.28$, $P < 0.001$) but similar incomes in other activities (Fig. 7.3.). Mean forest income was 13 times higher in rich than in poor households, corresponding to rich households' greater income from routine forest use (Kruskal-Wallis $x^2_2 = 20.87$, $P < 0.001$), but particularly malva nuts and timber (Kruskal-Wallis $x^2_2 = 26.55$, $P < 0.001$), relative to poor households. Rich households also sold significantly more forest products (70.9 ± 35.1%) than poor and medium-income households (35.9 ± 36.6% and 50.7 ± 31.0% respectively, Kruskal-Wallis $x^2_2 = 12.55$, $P = 0.002$). The income share of forest resources (all products) showed an apparent increase along the gradient of poor (29.0% of total income), medium-income (40.9%) and rich households (53.3%); however, income shares of only routine forest resources showed no such trend (34.3%, 19.3% and 29.7%

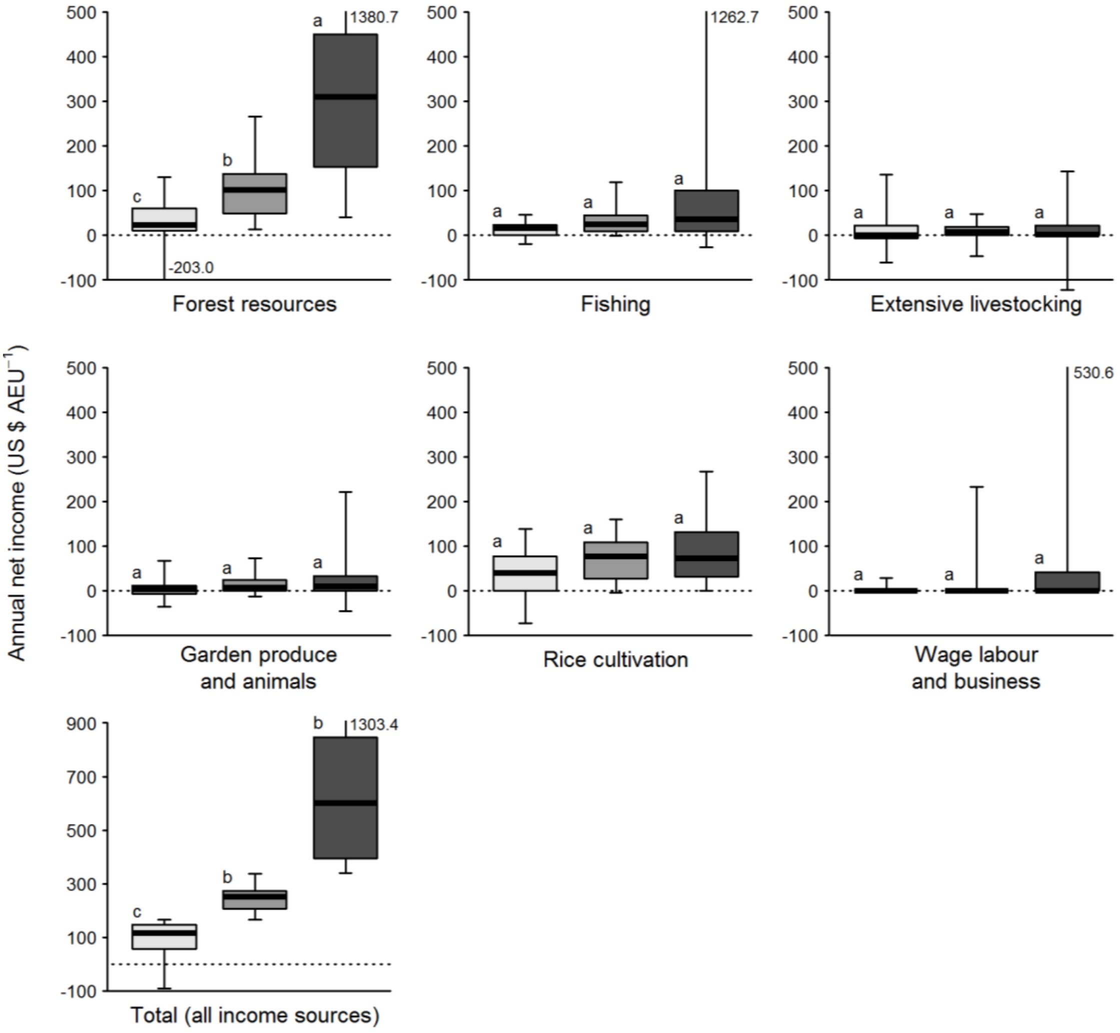

Figure 7.3. Net income from livelihood activities for a one-year period per household and by income tercile: poor (pale grey), medium-income (grey) and rich (dark grey) households. Thick horizontal bars indicate the median; minimum/maximum values beyond the axis range are labelled. Boxes sharing a common letter are not significantly different ($P < 0.002$) in Mann-Whitney tests with Bonferroni correction); see section 7.6.2 (Appendix B, Table 7.A3.) for test statistics.

respectively, excluding malva nuts and timber from both forest and total incomes). Results are consistent with and without AEU standardisation; poor households still only gained 11.1% of rich households' average forest income when calculated per household instead of per AEU (7.7% when AEU was used).

Households owned a mean of 7.1 ± 7.1 (± SD, $n = 64$) livestock, comprising 3.9 ± 3.5 buffalo and 3.1 ± 5.1 cattle. The capital value of households' herds averaged US$307.1 ± 289.2 per AEU with no difference between income terciles (Fig. 7.4.A, Kruskal-Wallis $x^{2}_{2} = 3.80$, $P = 0.149$). Livestock capital assets typically exceeded total livelihood incomes in poor and medium-income households (Fig. 7.4.B); livestock herd value in ten (53%) poor

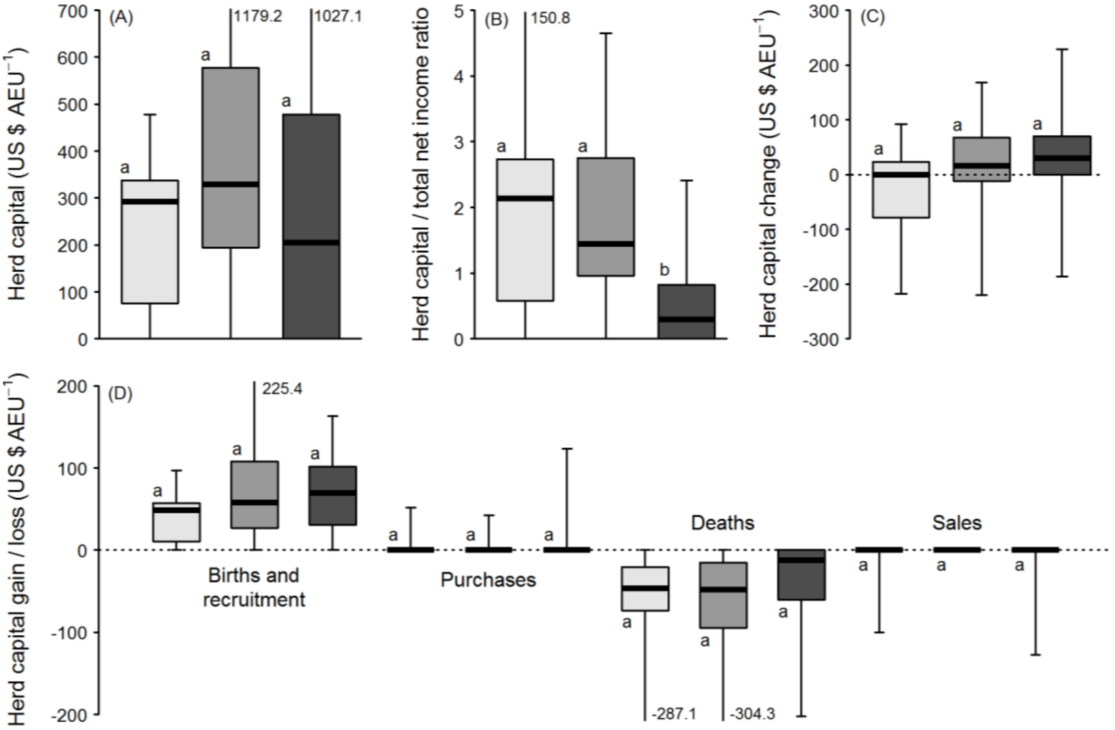

Figure 7.4. Total livestock herd capital per household (A); the ratio of herd capital relative to total income per household (B); change in herd capital (C); and the contribution of births and recruitment, animal purchases, deaths and animal sales to herd capital over the one-year study period and by income tercile: poor (pale grey), medium-income (grey) and rich (dark grey). Thick horizontal bars indicate the median, minimum/maximum values beyond the axis range are labelled. Boxes sharing a common letter are not significantly different ($P < 0.032$) in Mann-Whitney tests with Bonferroni correction, analysis in (D) was by four separate Kruskal-Wallis tests; see Appendix B (Table 7.A4.) for test statistics. (A) and (C) have $n = 64$, (B) excluded two households with negative income ($n = 62$), and (C–D) comprise livestock-owning households only ($n = 53$).

households was more than double total livelihood income. Livestock herd value was typically less than total income in rich households, but still equivalent to 57.4% of income on average. Herd capital change over the study year was variable in all income terciles (Fig. 7.4.C), with births, recruitment and deaths (rather than sales) contributing most to herd value change (Fig. 7.4.D). Animal purchases and sales in the year were rare (seven and three transactions respectively), involving 1.7 ± 0.8 animals per transaction and eight (15%) of 53 livestock-owning households. Reasons for selling livestock (in the household member's adult lifetime) were largely to support family welfare, living costs or investment in other capital assets (Fig. 7.5.). Obtaining cash to buy food and household provisions was most common; at least two households did this to compensate for a poor rice harvest.

7.4.2. Livelihood change

Agricultural mechanisation was demonstrated by accelerating purchases of hand-tractors in the years 2000–2010, with seven times more purchased in 2010 than 2005 (Fig. 7.6.). Sixty-three of 150 households (42%) owned hand-tractors in 2011, and 76.2% of these sold

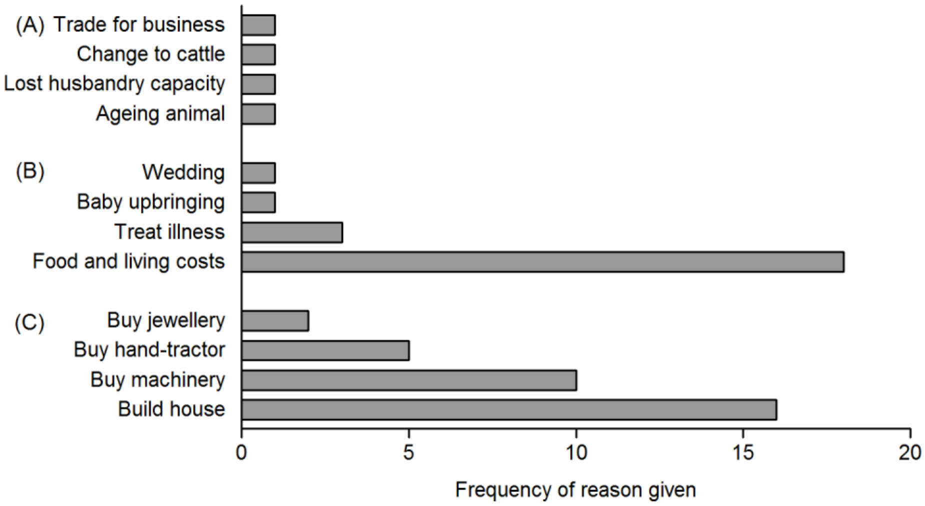

Figure 7.5. Reasons for selling livestock in households' lifetimes. Livestock sales were confirmed in 30 households and respondents were asked to give all the reasons for these sales. Responses relate to livestock husbandry matters (A), support of family members and living costs (B) and investment in other capital assets (C). "Machinery" includes boats, motorbikes and rice-milling machines.

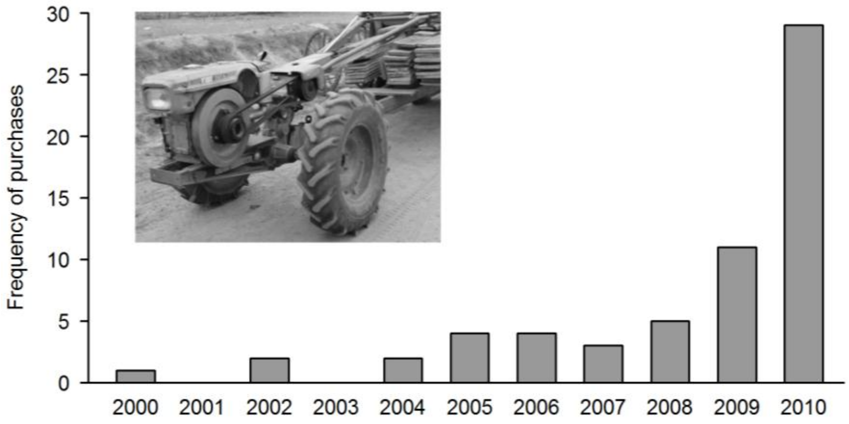

Figure 7.6. Frequency of hand-tractor (inset) purchases by year from a survey of 150 households.

livestock to assist with the purchase cost (US\$1919.4 ± 422.3, mean ± SD). Households selling livestock sold 43.2 ± 19.3% of their herds (5.5 ± 2.5 animals) comprising a similar number of buffalo (2.8 ± 2.0) and cattle (2.7 ± 2.5; Wilcoxon $V_{47,47} = 513$, $P = 0.837$). All 63 households owning hand-tractors used them for transport and 92.1% ploughed with them. The machines' greater draught capacity compared to livestock, and the latter's vulnerability to disease, were given as the main factors motivating hand-tractor purchases by key informants and focus group participants (Fig. 7.6.); disease (specified by a vet to include foot-and-mouth) was considered the biggest problem for livestock rearing by 7/9 informants.

Qualitative data suggested varied uses for livestock retained following hand-tractor purchases: 3/6 informants believed that hand-tractor owners continued using livestock for transport and ploughing but 3/6 believed they no longer fulfilled these functions. Approximately half of participants in the tractor-owner focus group still used livestock for these activities on an occasional basis when tractors broke down or they lacked money for running costs. Beyond draught, testimonials suggested that remaining livestock continued to fulfil other functions, including capital gain from herd growth. However, disease may influence households' decisions to keep livestock (Fig. 7.7.) and 5/6 informants believed buffalo numbers were declining while cattle were increasing.

Hand-tractor advantages and purchase motivations
FG1i: Hand-tractors are easier [than livestock] because you don't have to go and find them in the forest and they're much quicker and stronger. *(Tractor owner at a focus group)*
FG3i: When livestock die then that's it, can't do anything, but hand-tractors can be repaired when they break. They don't need looking after like the animals. *(Tractor owner at a focus group)*

I14: Some people look at their buffalo and if a lot are ill then they think it's better to sell them and get a hand-tractor instead […] the living thing is a greater risk […] and they could lose money. *(Chief of commune for 9 years, life-long local resident and livestock owner)*

Livestock use after hand-tractor purchases
I14: People […] may still use livestock when they don't have money for the petrol or repairs to their tractor. *(Chief of commune for 9 years and life-long local resident)*
FG3i: It's important to keep the females to get the calves, and to sell them to buy equipment or to pay for repairs when the tractor breaks. *(Tractor owner at a focus group)*
FG3iii: I don't use my ox or oxcart at all anymore […] if the tractor is broken I get it repaired quickly instead of using my oxen. *(Tractor owner at a focus group)*

I12: There were more buffalo in the past but several years ago lots of buffalo caught an infectious disease [...] some people lost all of their buffalo. *(Villager and livestock owner, part-time vet and 7 years conservation NGO experience working locally)*

FG2ii: I want to keep them [the livestock] to get the offspring [to gain a larger herd] but if the disease is really bad this year then I'll be forced to sell them. *(Tractor owner at a focus group)*

Figure 7.7. Testimonials from key informants and tractor-owner focus group participants on the advantages of investing in a hand-tractor and uses for livestock following hand-tractor purchases.

7.5. Discussion

7.5.1. Importance of livestock, rice and environmental income

Extensively-reared livestock (buffalo and cattle) were owned by 83% of households, but earned very little income relative to the diverse range of other livelihood activities (2.4% income share across households), a result echoed in other Cambodia forest communities (McKenney et al. 2004). Nonetheless, livestock provided a substantial capital asset, with herd value exceeding total livelihood income in 48% of households. Animal sales were made when households required finance to purchase other fixed capital, invest in a livelihood activity, or for times of particular need. The paucity of livestock sales observed may relate to their use in times of crisis rather than for regular trade, but high incomes from malva nuts and timber during the study year could also be a factor. Livestock may provide a form of insurance during livelihood shortfalls (Moll 2005), particularly in poor households where income deficits were 2.3 times more frequent (for any given livelihood activity) than in rich households. Although increases in herd capital over the year were typically small and only achieved by 57% of livestock-owning households, tractor-owners' hopes of breeding more livestock point to the potential for wealth accumulation, with livestock operating as a "living savings account" (Doran et al. 1979; Moll 2005).

Rice was cultivated by 84% of households and was the second most important income source (20.2% income share across households). Forest use was the only activity exceeding income from rice cultivation (48.1% income share), but much of forest income came from high-value malva nuts and timber, harvested at unusually high levels in the study period. Excluding these products illustrated likely household income in a more typical year, showing that rice cultivation and (routine) forest use may be of similar importance (the latter providing a 20.0% income share). The majority of rice cultivation's value was derived from households' own rice consumption, feeding people and home-reared livestock. Rice is a staple of the Cambodian diet (Nesbitt 1997) and a core livelihood component in many Cambodian forest communities (Clements et al. in press; McKenney et al. 2004).

Harvesting of forest products, fishing and extensive livestock rearing (reliant on natural graze available in the forest) were all common forms of environmental resource use; every household collected forest resources, 92% fished, and more than three-quarters kept large livestock. The combined environmental income from these activities (across 64 households) was 2.7 times greater than agricultural income (rice cultivation and garden produce), demonstrating the community's dependence on open-access resources in the dry forest landscape. The estimate may even be conservative, as respondents may have withheld data on illegally-collected products (e.g. timber and protected animals) and long recall periods can underestimate natural resource use (Lund et al. 2008). Environmental income remained considerable when only routine forest use was considered, becoming 1.5 times greater than income from agricultural activities. Environmental income is similarly important to other forest communities of Cambodia and Vietnam (Hansen & Neth 2006; McElwee 2008; McKenney et al. 2004) and throughout the developing world (Vedeld et al. 2007). However, while many studies observe inverse relationships between household wealth and forest use (e.g. Cavendish 2000; Rayamajhi et al. 2012), crude income shares in poor, medium-income and rich households showed no such a pattern in this community. Rather than rich households using the forest less, greater income and sale of forest resources (particularly high-value products such malva nuts and timber) was their defining feature, as incomes from other livelihood activities were similar across income terciles.

The farming practices key to White-shouldered Ibis survival also proved fundamental to local livelihoods: extensive livestock rearing creates grazed habitat for foraging ibis (Wright et al. 2010; chapter 6) and an important capital asset for local households, while rice cultivation provides wet-season foraging habitat for ibis (Fig. 7.1.B) and the second largest income source of dry forest livelihoods. Nevertheless, the overwhelming importance of environmental income to local people implies that the strongest link between livelihoods and the ibis is the shared use of dry forest resources and habitats. These results suggest that conservation could provide mutual benefit to people and ibis by safeguarding livelihoods and dry forest landscapes from imminent agricultural land acquisitions; however, livelihood change must also be considered before advocating this win-win strategy.

7.5.2. Evidence and impacts of livelihood change

Livelihood change and the modernisation of agricultural practices was demonstrated by the seven-fold increase in hand-tractor purchases from 2005–2010; similar mechanisation has been observed in Preah Vihear province, *c.* 100 km from the study site (Clements et al. in press). Hand-tractors provide greater speed, power and convenience than livestock, and these appeared to be major pull factors of mechanisation. As most purchases were recent and the study period was short, the impacts of mechanisation on livestock abundance were not observed directly. Nonetheless, more than three-quarters of tractor-owning households sold nearly half of their livestock, and more will be sold if tractor ownership continues to increase. Some tractor owners hoped to retain some livestock in case of breakdowns, financial shortfalls or to accrue herd capital, but whether growth from remaining stock can sustain livestock numbers is uncertain.

Tractor owners’ concerns for the health of their herds and anecdotal observation of people panic-selling buffalo during disease outbreaks (occurring since the study period; Bou Vorsak pers. comm.), suggest disease is another driver of livestock loss, directly killing animals but also encouraging further mechanisation. The risks of substantial capital-asset loss and livelihood shocks when livestock die, such as a lack of ploughing draught for rice cultivation (Shankar et al. 2012), may be an important push factor in hand-tractor purchases. Informants suggested buffalo numbers were declining and cattle were increasing; this may relate to a disproportionate impact of livestock disease on buffalo (agreed by 4/6 informants), as the ratios of cattle and buffalo sold to purchase hand-tractors were similar.

Evidence of change in other, more developed parts of central Indochina provides an indication of how livelihoods may progress; buffalo decline is a widespread phenomenon in Southeast Asia, with an 18.5% drop occurring from 1990–2002 (Nanda & Nakao 2003). Several decades ago, much of northeast Thailand was similar to the present-day study site, comprising forest-mosaic landscapes with communities reliant on low-intensity agriculture and forest access for resources and livestock grazing (Rigg 1993; Vityakon et al. 2004). From 1976 to 1997, agricultural modernisation (hand-tractors replacing livestock draught and chemicals replacing livestock manure) and loss of free grazing land contributed to a 36% decline of buffalo (Simaraks et al. 2003; Vityakon et al. 2004). The decline has eroded traditional uses of livestock, including as capital assets, leading to greater dependence on monetary institutions and consumer goods (Simaraks et al. 2003). This transition may now be beginning at this study site.

Livestock declines are likely to affect habitat availability for White-shouldered Ibis, given its need for grazed foraging habitats (Wright et al. 2010; chapter 6). Although optimal livestock densities are not yet known, substantial declines may be detrimental, as buffalo are now the main agent keeping waterholes open, providing the ibis's key foraging habitat in the breeding season (chapter 6). Other likely livelihood changes also create conservation concern. Market access – improved by a new main road – may create commercial opportunities such as cash cropping (e.g. Hamlin & Salick 2003; Thongmanivong & Fujita 2006), driving livelihoods away from traditional activities that support the ibis. In the longer term, human population growth will be substantial, as 63.5% of the study site population is < 25 years old (Ministry of Planning 2007). Population effects on agriculture are complex and contested (Lambin et al. 2001), but increasing pressure on land to provide food (or cash with which to purchase it) may drive greater land use for agriculture and/or higher-intensity production, creating an uncertain future for the ibis. Despite the shared importance of livestock rearing, rice cultivation and the dry forest landscape to local people and White-shouldered Ibis, changing livelihoods look set to imminently uncouple this linkage. Traditional farming practices could become increasingly uneconomic or socially inappropriate as agriculture is modernised, creating not a win-win scenario but a trade-off between local development and ibis conservation.

7.5.3. The conservation approach

White-shouldered Ibis is amongst a distinct group of developing-world species depending on traditional farming systems (Wright et al. 2012b; chapter 2). In the absence of the natural processes that historically sustained these species, maintaining or mimicking traditional practices is a conservation priority. Win-win strategies may appeal where wildlife and people share a reliance on farming practices or a common threat, and where traditional livelihoods appear stable (at least in the medium term). However, where access to new markets and technology is expected within years rather than decades, traditional agriculture may be rapidly modernised and win-win strategies may be ephemeral. The need to increase agricultural production, driven by population growth and greater affluence (Godfray et al. 2010), may exert further pressure on some of these farming systems. In these contexts, new conservation mechanisms are required to continue managing landscapes for farming-dependent species. Possibilities for incentivising local farmers include direct payments for conservation, payments for environmental services and market-enhancing certification schemes (Ferraro & Kiss 2002), but in maintaining traditional livelihoods, the opportunity costs to local development will need explicit consideration.

7.6. Appendices

7.6.1. **Appendix A**: Inputs and outputs of household net incomes

Activity	Intermediate inputs / capital costs	Outputs
Forest resources	Axe purchase Chainsaw purchase/running cost Hand-tractor purchase/running cost Motorbike purchase/running cost Own ox-cart draught used (+L) Other (e.g. push-bike purchase)	Construction timber [a] Illegal timber [a] Grass for roof construction Bamboo poles Fence poles Fuelwood Rattan Tree resin Wild mammals and reptiles Wild amphibians Malva nuts [a] Other wild fruits and vegetables
Fishing	Net purchase/repair Boat repair Hand-tractor purchase/running cost Motorbike purchase/running cost Own ox-cart draught used (+L)	Small fish and eels Large fish and eels Fermented fish paste
Extensive livestock rearing [b]	Buffalo/cattle purchases Rice crop straw (+R) Ox-cart draught hired [c] Ploughing draught hired [c] Medical treatment Vet service	Buffalo/cattle sales Own ox-cart draught used Own ox-cart draught let-out Own ploughing draught used Own ploughing draught let-out Ox-cart/ploughing draught given [c]

Table 7.A1. Continued pg. 151.

Garden produce and animals	Pig/chicken/duck purchases Crop seed bought Rice dust (+R) Cooked rice (+R) Tool purchase/repair Medical treatment Vet service	Pig/chicken/duck meat/sales Duck eggs Fruit and vegetables Maize crop [d] Sugar cane crop [d] Crop seed [d]
Rice cultivation	Rice seed purchase Plough material purchases Hand-tractor purchase/running cost/hire Own ox-cart draught used (+L) Own ploughing draught used (+L) Ox-cart/ploughing draught given (+L) Rice threshing service bought Rice mill service bought Labour hired	Rice crop [d] Rice dust Rice crop straw Cooked rice for animals Rice mill service let-out Rice seed [d]

Table 7.A1. Summary of major inputs and outputs contributing to the net income of five key livelihood activities valued at 64 households for a one-year period. (+L) indicates products also booked as outputs of extensive livestock rearing, (+R) indicates products also booked as outputs of rice cultivation. For wage labour and business respondents stated their salaries or estimated the profit made from business activities.

[a] Key informant evidence demonstrated these products were collected in atypically high quantities during the study year, and that the vast majority of input costs to forest resource collection resulted from collection of these products. Net income from more routine forest use (comprising the other forest resource outputs) was calculated with the assumption that chainsaws, hand-tractors, ox-cart draught and other inputs were only used for timber and malva nut collection. Net income from routine forest use is therefore a best-estimate rather than a precise value.

[b] In contrast to other grazing systems (Babulo et al. 2009; Narain et al. 2008), livestock fodder was not collected from the forest and so no market price or contingent value was available; fodder value is therefore not accounted for in extensive livestock rearing and forest resource use activities *per se*. However, the value of fodder is indirectly represented by income from extensive livestock rearing as a whole, and this is incorporated in total environmental income. Livestock manure was also not collected and quantifying the value of manure from animals kept haphazardly on rice fields was beyond the scope of the study.

[c] To avoid double counting, ox-cart and ploughing draught hired for any of the six livelihood activities was accounted as an input to extensive livestock rearing and not to the activity that

required its use. Ox-cart and ploughing draught given to the household was accounted as an input to the activity that required it and an output of extensive livestock rearing.

[d] Crop harvests (namely rice) were sometimes in the middle of the study year, therefore crop and seed use within the study period was inevitably from cultivation seasons both during and prior to the study. To prevent erroneous counting of outputs from multiple harvests, only crops and seeds used during the study year (originating from either during or prior to study) were included, those harvested but unused (e.g. remaining in storage for use in the next season) were not counted.

7.6.2. **Appendix B**: Test statistics

	Forest	Fishing	Livestock	Garden	Rice
Fishing	$V = 1791$ $P < 0.001$	–	–	–	–
Livestock	$V = 1892$ $P < 0.001$	$V = 1446$ $P = 0.005$	–	–	–
Garden	$V = 1969$ $P < 0.001$	$V = 1523$ $P = 0.006$	$V = 743$ $P = 1.000$	–	–
Rice	$V = 1557$ $P = 0.008$	$V = 439$ $P = 0.004$	$V = 184$ $P < 0.001$	$V = 1590$ $P < 0.001$	–
Wage	$V = 1840$ $P < 0.001$	$V = 1465$ $P = 0.027$	$V = 842$ $P = 1.000$	$V = 1304$ $P = 1.000$	$V = 1038$ $P = 0.008$

Table 7.A2. Statistics for paired Wilcoxon tests ($V_{64,64}$) of pairwise differences in livelihood activity net incomes (US$ per adult equivalent unit), with Bonferroni adjustment (see Fig. 7.2. above). These post-hoc tests follow a Friedman test of livelihood activity net incomes (Friedman $x^2_5 = 95.52$, $P < 0.001$). “Forest” = forest resource use. “Livestock” = extensive livestock rearing. “Garden” = garden produce and animals. “Wage” = wage labour and business.

(a) Forest resource use

Kruskal-Wallis $x^2_2 = 35.28$, $P < 0.001$

	Poor	Medium
Medium	$W = 82.5$, $P = 0.002$	–
Rich	$W = 21$, $P < 0.001$	$W = 65$, $P < 0.001$

(b) Fishing

Kruskal-Wallis $x^2_2 = 6.88$, $P = 0.032$

	Poor	Medium
Medium	$W = 134$, $P = 0.091$	–
Rich	$W = 139$, $P = 0.079$	$W = 195$, $P = 1.000$

(c) Extensive livestock rearing

Kruskal-Wallis $x^2_2 = 0.24$, $P = 0.886$

	Poor	Medium
Medium	$W = 206$, $P = 1.000$	–
Rich	$W = 212.5$, $P = 1.000$	$W = 241$, $P = 1.000$

(d) Garden produce and animals

Kruskal-Wallis $x^2_2 = 3.00$, $P = 0.223$

	Poor	Medium
Medium	$W = 160$, $P = 0.131$	–
Rich	$W = 170.5$, $P = 0.430$	$W = 226$, $P = 1.000$

(e) Rice cultivation

Kruskal-Wallis $x^2_2 = 4.61$, $P = 0.099$

	Poor	Medium
Medium	$W = 156.5$ $P = 0.330$	–
Rich	$W = 149$, $P = 0.140$	$W = 209$, $P = 1.000$

(f) Wage labour and business

Kruskal-Wallis $x^2_2 = 5.89$, $P =0.061$

	Poor	Medium
Medium	$W = 179$, $P = 0.270$	–
Rich	$W = 166$, $P = 0.063$	$W = 201$, $P = 1.000$

(g) Total net income

Kruskal-Wallis $x^2_2 = 56.00$, $P < 0.001$

	Poor	Medium
Medium	$W = 441$, $P < 0.001$	–
Rich	$W = 462$, $P < 0.001$	$W = 462$, $P < 0.001$

(h) Forest timber and malva nuts

Kruskal-Wallis $x^2_2 = 26.45$, $P < 0.001$

	Poor	Medium
Medium	$W = 363$, $P = 0.001$	–
Rich	$W = 422$, $P < 0.001$	$W = 328$, $P = 0.057$

(i) Routine forest use

Kruskal-Wallis $x^2_2 = 20.87$, $P < 0.001$

	Poor	Medium
Medium	$W = 253$, $P = 1.000$	–
Rich	$W = 391$, $P < 0.001$	$W = 392$, $P < 0.001$

Table 7.A3. Statistics for Kruskal-Wallis and post-hoc Mann-Whitney tests (*W*, with Bonferroni adjustment) of differences in net income (US$ per adult equivalent unit) per income tercile (poor, medium-income and rich households) for each livelihood activity (a–f), total net income (g; see Fig. 7.3. above). Comparison of malva nut and timber income (h) and routine forest income (i), subsets of overall forest use (a), are also included. Poor households $n = 21$, medium-income households $n = 21$ and rich households $n = 22$.

(a) Livestock herd capital

Kruskal-Wallis $x^2_2 = 3.80$, $P = 0.149$

	Poor	Medium
Medium	$W = 143$, $P = 0.160$	–
Rich	$W = 223$, $P = 1.000$	$W = 287.5$, $P = 0.520$

(b) Herd capital/total net income ratio

Kruskal-Wallis $x^2_2 = 12.66$, $P = 0.002$

	Poor	Medium
Medium	$W = 197$, $P = 1.000$	–
Rich	$W = 307$, $P = 0.031$	$W = 373$, $P = 0.002$

(c) Herd capital change

Kruskal-Wallis $x^2_2 = 2.35$, $P = 0.308$

	Poor	Medium
Medium	$W = 122.5$, $P = 0.670$	–
Rich	$W = 104$, $P = 0.500$	$W = 153.5$, $P = 1.000$

(d) Livestock births and recruitment

Kruskal-Wallis $x^2_2 = 2.97$, $P = 0.227$

	Poor	Medium
Medium	W 114 =, $P = 0.400$	–
Rich	W 102 =, $P = 0.440$	$W = 166$, $P = 1.000$

(e) Livestock purchases

Kruskal-Wallis $x^2_2 = 3.25$, $P = 0.197$

	Poor	Medium
Medium	$W = 190$, $P = 0.420$	–
Rich	$W = 168$, $P = 0.600$	$W = 160$, $P = 1.000$

(f) Livestock deaths

Kruskal-Wallis $x^2_2 = 2.68$, $P = 0.262$

	Poor	Medium
Medium	$W = 151$, $P = 1.000$	–
Rich	$W = 182$, $P = 0.580$	$W = 207.5$, $P = 0.430$

(g) Livestock sales

Kruskal-Wallis $x^2_2 = 2.20$, $P = 0.332$

	Poor	Medium
Medium	$W = 180.5$, $P = 0.420$	–
Rich	$W = 152$, $P = 1.000$	$W = 152$, $P = 0.950$

Table 7.A4. Statistics for Kruskal-Wallis and post-hoc Mann-Whitney tests (W, with Bonferroni adjustment) of differences in livestock herd capital (a); ratio of herd capital to total net income (b); herd capital change in the one-year study period (c); capital gain from livestock births and recruitment (d); capital gain from livestock purchases (e); capital loss from livestock deaths (f); and capital loss from livestock sales (g), per income tercile (poor, medium-income and rich households; see Fig. 7.4. above). All capital values were in US$ per adult equivalent unit. Poor households $n = 21$, medium-income households $n = 21$ and rich households $n = 22$.

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Chapter 8

Experimental test of a conservation intervention for a highly threatened waterbird

Local nest guardian and camp, stationed near a White-shouldered Ibis nest.

8.1. Abstract

Human exploitation and disturbance often threaten nesting wildlife. Nest guarding, a technique that employs local people to prevent such interference, is being applied to an increasing number of species and sites, particularly in South-East Asia. Although recent research has begun to assess nest guarding cost-effectiveness, case-control studies are rare and the circumstances in which the schemes are most useful remain unclear. The study experimentally tested the effect of nest guarding for the Critically Endangered White-shouldered Ibis *Pseudibis davisoni*, a species exploited opportunistically for food and now largely confined to dry forests in Cambodia. A randomised sample of 24 and 25 nests were guarded and unguarded, respectively, at a single site over two years. To assess the potential for nest guarding in different conservation contexts, the effect of conservation activity as a whole was investigated by comparing ibis nest failure at four sites with different intervention histories. Nest guarding had no detectable effect on nest success at the principal site. Across all sites, nest failure varied by up to 63% but an effect of conservation activity was not found; nest failure was actually increased by conservation activity at one site, indicating the need for careful implementation of guarding schemes that benefit only a small number of people in the local community. Comparison with other studies suggests nest guarding effectiveness is likely to be context specific and may differ between species that are exploited opportunistically, such as White-shouldered Ibis, and those routinely targeted for trade.

8.2. Introduction

Improving nest success is a fundamental conservation measure for many threatened species including birds (Bell & Merton 2002; Jones 2004) and reptiles, particularly turtles (Spotila 2004). Nests fail for a variety of reasons including human exploitation, disturbance and predation. Anthropogenic nest failures may be substantial where people utilise nest contents for food or trade (Tomillo et al. 2008; Wright et al. 2001). Nest exploitation and destruction can be mitigated through various interventions, including awareness campaigns (Barré et al. 2010; Herrera & Hennessey 2007); conservation payment schemes, often for nest guarding (Clements et al. 2010; Niesten & Gjertsen 2010); harvesting quotas (Hobbs 2004); law enforcement (Cahill et al. 2006); or a combination of these (Boussekey 2000).

Nest guarding schemes employ local people to deter human interference at nests. Guard salaries provide an incentive to report a nest site and ensure its success, rather than harvesting its contents. The approach is becoming popular, particularly for the protection of sea turtle colonies (Ferraro & Gjertsen 2009) and, in South-East Asia, for the protection of waterbirds (Clements et al. 2010; Sok et al. 2012), parrots (Widmann & Widmann 2008), raptors (Prawiradilaga 2006; Salvador & Ibanez 2006) and a bustard (Packman 2011). These species are vulnerable because of their conspicuousness, value or mere proximity to rural communities depending on natural resources. In Cambodia, for example, twelve threatened birds and one river turtle species are currently protected by nest guarding (Clements et al. 2010; Packman 2011; Sok et al. 2012).

Evaluating the effectiveness of interventions is important to ensure that they: achieve desired goals; do not inadvertently increase problems – for example disturbance-induced nest failure associated with nest searching and guard presence; and represent efficient use of resources and time (Ferraro & Pattanayak 2006; Sutherland et al. 2004). Although studies have begun to assess the effectiveness of nest guarding (e.g. Clements et al. in press; Ferraro & Gjertsen 2009) this intervention is very rarely tested experimentally. Unlike other nest protection interventions (Keo et al. 2009; e.g. Kragten et al. 2008), nest guarding schemes are typically implemented across all monitored nests, leaving no unprotected nests as a control treatment; evaluation has therefore frequently depended on population trend data that can be confounded by other factors, such as weather, fluctuation in predator populations or other conservation activities.

Recent studies of nest guarding effectiveness in Cambodia have shown contrasting results. Sok et al. (2012) found little effect with three waterbird species, although no control was included. However, Clements et al. (in press) used a quasi-experimental method that matched guarded nests in protected areas with unguarded nests in unprotected areas, determining that nest guarding (perhaps in combination with other interventions and their impact on community behaviour) successfully increased nest success for two waterbird species. Clearly the issue requires further study.

The Critically Endangered White-shouldered Ibis *Pseudibis davisoni* is one of South-East Asia’s most threatened waterbirds (Tordoff et al. 2005). Using this as a model species, the study tested the independent effect of nest guarding using a case-control design – the first randomised experimental test of this intervention. To understand further the circumstances in which nest guarding may be effective, the rates and causes of nest failure were also examined across four ibis subpopulations with different conservation histories, investigating the relative contribution of human interference, and the value of conservation activity (encompassing a broad range of interventions) in improving nest success. However, potentially confounding ecological and social factors, and small sample sizes, render this aspect of the study preliminary. As the number and scale of nest guarding schemes look set to increase, this study provides conservation practitioners with new evidence regarding this intervention's effectiveness.

8.3. Methods

8.3.1. Study species

White-shouldered Ibis historically occurred across Indochina, but in the twentieth century the population severely contracted so that 85–95% of the world's remaining 731–856 birds are now found in Cambodia (Wright et al. 2012; chapter 3). The species shares its dry forest habitat with human communities dependent on natural resources and, although not valued for trade, is exploited opportunistically for food (Sok et al. 2012; HLW pers. obs.). It is a solitary, dry-season breeder (December–May), building nests in tree canopies typically 10–25 m above the ground. Nesting most frequently occurs in open deciduous dipterocarp forest, forest remnants or isolated trees at rice fallows (Clements et al. in press; HLW unpubl. data) or in seasonally flooded forest along large rivers (Sok et al. 2012; Sutrisno et al. 2009).

8.3.2. Study sites

Western Siem Pang Important Bird Area (Seng et al. 2003) in Stung Treng province (14°07'N 106°14'E) was the principal site and location of the nest guarding experiment. This 138,000 ha dry forest landscape holds the largest known population of White-shouldered Ibis, at least 226 birds (Wright et al. 2012; chapter 3). Approximately 11,000 people (Ministry of Planning 2007) live in small settlements concentrated in the centre and east of the site. Small-scale conservation action has taken place continuously since 2003 with three local staff and a project officer dedicated to waterbird interventions and monitoring (BirdLife International 2009). A member of the local Forestry Administration has led the team in law enforcement, tackling exploitation of threatened waterbirds; two police officers joined the team in 2011.

White-shouldered Ibis nest failure was compared among four study sites (including Western Siem Pang) along a gradient of conservation activity, varying from multiple measures applied over several years, to recent, small-scale conservation with few interventions.

Conservation measures included: ecotourism; nest finding reward schemes; nest guarding; education and awareness campaigns; community- and ranger-based biodiversity monitoring; ranger- or police-based law enforcement; and agricultural certification schemes (BirdLife International 2009; Clements et al. 2010). Using the year that waterbird conservation began, sites were ranked by duration of conservation activity as a proxy for the degree of intervention, as follows:

1) Kulen Promtep Wildlife Sanctuary, Preah Vihear province (13°58'N 104°53'E), is a protected area containing a small subpopulation of White-shouldered Ibis in a dry forest landscape (Wright et al. 2012; chapter 3). Waterbird conservation began in 2002, and is undertaken by approximately seven staff. Conservation activity is well developed, with local people participating in ecotourism, nest finding, nest guarding (including eight bird species) and an agri-environment scheme (Clements et al. 2010; Clements et al. in press). Biodiversity monitoring and law enforcement have involved teams of rangers since 2004 and 2006 respectively; the latter has targeted waterbird exploitation amongst other illegal activities. This site also forms part of Clements et al.’s (in press) nest guarding study, but involving different study species.

2) Western Siem Pang, ranked as the site with second-longest waterbird intervention.

3) The Mekong Flooded Forest (henceforth Mekong), tracking the Mekong River between Kratie and Stung Treng towns (13°02'N 106°01'E), has globally significant ibis numbers (Timmins 2008; Wright et al. 2012; chapter 3) and a short history of conservation activity. Local conservation awareness remains low (Sok et al. 2012) but two staff coordinated nest finding, monitoring and guarding of three bird species for two years, in 2008–09 and 2010–11. In 2008–09 guarding was less intensive than at other study sites, as guards were not present at nests during all daylight hours, but this improved in 2010–11 (Sok et al. 2012). Law enforcement has comprised one Forestry Administration officer tackling exploitation of threatened waterbirds in 2008-9 and 2010-11.

4) Lomphat Wildlife Sanctuary, Ratanakiri province (13°20'N 106°56'E), has the second-highest known population of White-shouldered Ibis (Wright et al. 2012; chapter 3). Law enforcement by rangers has occurred since 2003 but with low priority given to threatened waterbirds. Waterbird conservation measures were not adopted until 2010, when four staff began nest finding, monitoring and patrolling in cooperation with local communities (BirdLife International 2010). Extensive awareness campaigns and nest protection activities await implementation.

With three sites comprising dry forest landscapes and one dominated by braided riverine channels, environmental conditions are variable and contrasting ecology may contribute to differences in nesting success. Nevertheless, all sites were predominantly lowland areas with large expanses of deciduous dipterocarp forest containing patches of other habitat (semi-evergreen and mixed deciduous forests, open grasslands, active and abandoned rice paddies). Social context (such as settlement age) and livelihood strategies (such as dependence on fishing) may also differ between sites, but available evidence suggests that local motivations for White-shouldered Ibis nest exploitation (driven by consumption rather than trade) are similar (Sok et al. 2012; HLW pers. obs.) and nest robbery or destruction has been reported at all sites. A complete control site, lacking any intervention or influence of people at nests, was not available as no site was free from the influence of local communities or conservation activity.

8.3.3. Nest finding

Nests were located and monitored for three breeding seasons (2008–2011) at Western Siem Pang, the primary study site. Elsewhere, nests were monitored for the same three breeding seasons at Kulen Promtep, two seasons at the Mekong (2008–09 and 2010–11) and one season at Lomphat (2010–11). Nest sample sizes were constrained by the scarcity of the study species and few known nests (which are frequently occupied in successive years; HLW unpubl. data) prior to study inception. Nest reward schemes, applied at all sites, overcame this by providing a small cash incentive for local people to report nests. Additional active searching was conducted at old nest sites and new localities where ibis pairs were seen regularly. Searches were systematic in Western Siem Pang, where four staff worked full time. As nest sites became known to the staff the contribution of reward-scheme informants fell from 91% of nests in 2008–09 to 40% in each of the subsequent two breeding seasons. Staff search effort was less intensive and often opportunistic at the other three sites, reflecting lower capacity and/or other conservation priorities. Local people contributed 67% of nest finds at Kulen Promtep, 89% at Lomphat and 100% at the Mekong across all years. Differences in White-shouldered Ibis density and knowledge of nest locations resulted in contrasting nest sample sizes across study sites, and may have reduced the accuracy of nest failure estimates at three sites. The distances from nests to the nearest settlement did not differ ($F_{3,96} = 1.79$, $P = 0.154$) between the four study sites, indicating similar proximity to people.

8.3.4. Nest monitoring

Field staff were trained to monitor nests consistently at every site, recording nest activity and overall outcome. Nest guards were also trained to monitor nests twice daily; their records were corroborated by field staff observations. Monitoring frequency depended on site capacity; visits were typically every 5–7 days at Western Siem Pang, every 7 days at Kulen Promtep, and every 3–14 days at the Mekong and at Lomphat. Each monitoring visit lasted until nest status was identified – usually 30–60 minutes. If no ibis activity was observed after an hour, staff searched under the nest for evidence of failure. For purposes of analysis, causes of failure were determined from tangible evidence only and in circumstances of near or absolute certainty. In cases of scant evidence or subjective assessment by observers the cause of failure was considered unknown. Anthropogenic failures were indicated by climbing equipment or felling of the nest tree, but use of slingshots was undetectable. Reports of nest destruction were accepted when based on multiple sources or admissions by those responsible. Predation was only recorded when the event was actually observed.

The impact of high winds was inferred with medium-high probability, using knowledge of recent weather and likely susceptibility given the nest's location in the tree canopy. Premature flight, triggered by unknown causes, was assumed when near-fledged chicks were found dead beneath the nest with no evidence of predator damage. Partial brood loss was recorded opportunistically but its prevalence may have been underestimated, as initial clutch size could not be determined by ground-based observations. The degree to which different causes of nest failure were detected or under-recorded may have varied slightly among sites, owing to differences in capacity and frequency of nest visitation. The attributed causes of nest failure are therefore treated as indications of potential contributory factors rather than as an accurate measure of their importance. Predation and human disturbance were most likely to have remained undetected relative to other causes of failure.

8.3.5. Nest guarding

Nest guarding was implemented experimentally in the 2009–10 and 2010–11 breeding seasons at Western Siem Pang. Guarded and unguarded (control) treatments were randomly applied to 24 and 25 nests respectively, with guards recruited from local communities. Guards discouraged illegal exploitation or disturbance by threatening to report the perpetrators to the local Forestry Administration, but did not intervene in natural events such as predation. Guards and field staff remained concealed and at least 100 m from the nest to avoid their presence becoming a source of disturbance. Fourteen nests were -protected by a single guard each; guards started work within one to three days of nest discovery (depending on availability), were present during daylight hours and paid US$3.75 per day. Site inaccessibility and/or limited transport availability dictated that guards had to camp in the vicinity of the other 10 guarded nests; these had two guards to cover the logistics of camping and guarding (each paid US$4.25 per day), and may therefore have received greater protection than single-guard nests. Camps were at a sufficient distance from nests to prevent extra disturbance to nests or to potential nest predators. Distance of nests to nearest settlement was marginally less at guarded (range 0.48–9.96 km from settlement, mean 3.7 km ± 2.7 SD) than unguarded (range 0.12–10.84 km, 5.6 km ± 3.5, $t_{32.9} = -1.99$, $P = 0.054$) nests.

8.3.6. Analysis of nest survival

Western Siem Pang data from the 2009–10 and 2010–11 breeding seasons were modelled to determine the effectiveness of guarding and the predictors of nest failure. Data quality was sufficient to model nesting stages separately for (1) the combined incubation and chick-brooding stage, when the nest was almost constantly attended by at least one adult, and (2) the late-nestling stage, when both adults stopped sitting or crouching over chicks and were often absent together. These stages were chosen for two reasons. First, hatching date at some nests was not reliably determined from ground-based observations until chicks were large enough to be visible or adult behaviour changed, so that nests failing close to this date could not reliably be assigned as an egg- or chick-stage failure. Second, it is assumed that failure may be affected by chick size, adult ibis presence at nests and frequency of provisioning.

Nest outcome was considered in logistic regression models to predict daily failure rate (DFR), including the number of exposure days (that the nest was active and monitored) as the number of binomial trials (Aebischer 1999). The first round of modelling tested the effects of guarding (guarded/unguarded nests) and breeding season on DFR in each nesting stage. In the second round, guarding, breeding season and distance to settlement (square-root transformed to reduce leverage) underwent model selection for the incubation and brooding stage. Alternative models were evaluated by Akaike Information Criterion corrected for small sample size (AICc). Model selection was not undertaken for the late-nestling stage as only one failure occurred. Nesting date, measured as the number of days since breeding started (the earliest date that incubation was observed across all nests) to the date of brooding completion (or failure if earlier) at each nest, was examined during preliminary analysis but was a poor predictor and not considered further. The relative importance of variables was indicated by model-averaged parameter estimates (Burnham & Anderson 2002) and change in model AICc when terms were iteratively dropped from the best model; an increase in AIC of ≥ 2 units indicated strong support. Overall probability of nest success was calculated using estimated DFR.

A sensitivity analysis was undertaken to determine the minimum detectable effect size of guarding with the sample size achieved. Nest outcome and exposure day data were remodelled ten times, each time with a dummy treatment variable that comprised two randomly allocated treatment levels and created samples of 24 guarded and 25 unguarded nests (matching the experiment). Overall nest success confidence intervals (CI), derived from DFR estimates and averaged across the ten iterations, demonstrated the boundaries of a random null effect that were used to calculate the percentage difference in overall success required to detect an effect of guarding at $\alpha = 0.05$ for the incubation and brooding stage.

To compare nest failure prior to and during the nest guarding experiment, data for the incubation and brooding stage were pooled from the 2009–11 seasons and compared in logistic regression models with nests from the 2008–09 (when all nests were unguarded); models included terms for time period, and both time period and guarding (guarded/unguarded nests).

Nest sample sizes were smaller at the remaining three study sites than at Western Siem Pang, but sufficient to make a preliminary comparison of nest failure along the gradient of conservation activity. DFR per site was predicted using logistic regression models (as above), with nest success modelled across the entire nesting period (incubation to fledging) as data from many nests were not sufficient to distinguish individual nest stages. Breeding season was included as a categorical fixed factor to account for uneven sample sizes by site and season. Pairwise comparisons of failure rate between sites were made by changing the reference level of the site variable in the model. The ranking of study sites by conservation activity was related to site-specific DFR estimates – weighted by total exposure days and averaged across seasons – by non-parametric correlation with one-tailed probability.

8.4. Results

8.4.1. Causes of nest failure

A total of 100 White-shouldered Ibis nests, 33 of which failed, were monitored across the four study sites over three breeding seasons (2008–2011). Causes of nest failure remained unknown at 19 (58%) nests. Anthropogenic factors accounted for at least nine (27%) failures, involving nest robbery (4 nests) and, at the Mekong, envy-driven reprisals (5 nests) for the financial benefits received by nest guards. Strong winds were probably responsible for three further failures (9%). Premature flight by near-fledged chicks caused one failure and at least one partial brood loss, although what triggered chicks to bail remained unknown. Natural predation was confirmed in one complete nest failure, when a Southern Jungle Crow *Corvus macrorhynchos* removed all eggs of a clutch in the absence of adult ibises, and one partial loss of a further brood, when this species predated a newly hatched chick. Nocturnal predation could not be detected using this study's methodology and could have contributed to failures where the cause was unknown. Nest stage durations and the number of fledglings are given in the appendix, section 8.6.

8.4.2. Predictors of nest survival and the guarding effect

The nest guarding experiment comprised 49 nests over two breeding seasons in Western Siem Pang and nest guard salary payments totalled US$5,903. Only one failure was observed in the late-nestling stage, resulting in lower DFR (over both breeding seasons) than in the incubation and brooding stage (Table 8.1.). Estimated DFR was similar between nests with and without the guarding treatment (β = –0.25, CL ± 1.14 at guarded relative to unguarded nests) and between breeding seasons (β = 1.12, CL ± 1.52 in 2010–11 relative to 2009–11) in the incubation and brooding stage (Table 8.1., Fig. 8.1.). Overall success in this nesting stage was only 4.5% greater at guarded than unguarded nests in 2009–10, and 14.4% in 2010–11, compared to a minimum detectable effect of 33.5% (for $\alpha = 0.05$) given the study's sample size. Failure rate also did not differ (in a univariate model) with level of nest protection (β = –0.14, CL ± 1.70 for nests protected by two guardians relative to nests protected by one). There was no difference in DFR between time periods during and prior to the guarding experiment (β = –0.73, CL ± 1.04 for 2009–11 relative to 2008–09); with guarding also included in this model, there remained no effect of time period and no difference between guarded/unguarded treatments (β = –0.64, CL ± 1.20 at guarded relative to unguarded nests).

Nest stage	Season	Guarding	Nests	Exposure days	Failures	DFR	DFR 95% CI	Overall nest success	Nest success 95% CI
Incubation and brooding	2009–10	Guarded	9	320	1	0.0035	0.0000–0.0086	0.858	0.685–1.000
		Unguarded	5	196	1	0.0045	0.0000–0.0113	0.821	0.608–1.000
	2010–11	Guarded	15	455	5	0.0106	0.0017–0.0196	0.627	0.421–0.930
		Unguarded	14	370	5	0.0136	0.0024–0.0249	0.548	0.331–0.901
ΔAICc	0.284	–2.124							
Late-nestling	2009–10	Guarded	8	185	1	0.0054	0.0000–0.0159	0.873	0.667–1.138
		Unguarded	10	128	0	0.0000	0.0000–0.0000	1.000	1.000–1.000
	2010–11	Guarded	10	280	0	0.0000	0.0000–0.0000	1.000	1.000–1.000
		Unguarded	9	211	0	0.0000	0.0000–0.0000	1.000	1.000–1.000
ΔAICc	–0.534	–1.325							

Table 8.1. Estimates of daily failure rate (DFR) and probability of overall success of White-shouldered Ibis nests at Western Siem Pang. Incubation and brooding (model AICc = 76.19, dispersion ratio = 1.17) and late-nestling stages (model AICc = 13.39, dispersion ratio = 0.13) were modelled separately, each containing terms for breeding season and guarding (guarded/unguarded). ΔAICc is the change in AICc when the term is removed from the model (negative values indicate an improvement in model fit without the terms).

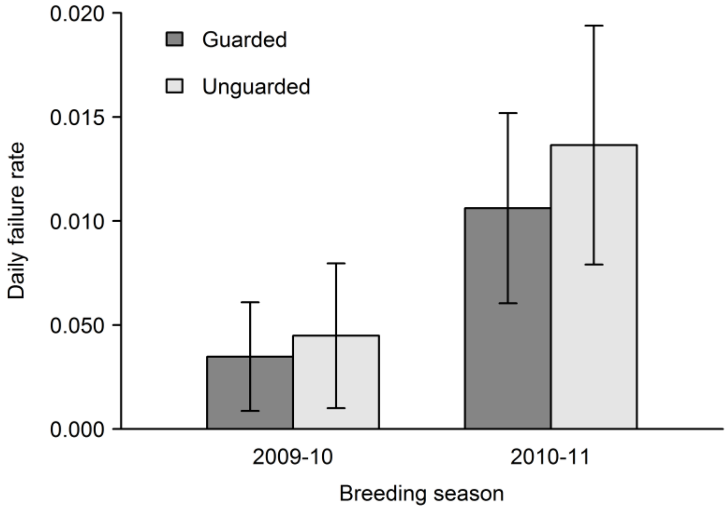

Figure 8.1. Daily failure rates of guarded and unguarded nests during the incubation and brooding stage, by breeding season. Daily failure rates were estimated using a binomial logistic regression model of nests at Western Siem Pang (Table 8.1.); error bars indicate standard errors.

Model #	Guarding	Distance to settlement	Breeding Season	AICc	ΔAICc	Akaike weight
6		▨	▨	72.12	0.00	0.36
2		▨		73.11	0.99	0.22
3			▨	74.07	1.95	0.14
7	▨	▨	▨	74.29	2.17	0.12
4	▨	▨		75.26	3.14	0.07
5	▨		▨	76.19	4.07	0.05
1	▨			76.47	4.35	0.04
Model averaged β	0.038	0.656	0.818			
95% CI	–0.356–0.433	0.001–1.312	–0.202–1.839			

Table 8.2. Multi-model inference and model averaging of nest failure models using Western Siem Pang nest data. Akaike Information Criteria (AICc) and Akaike weights are given for each candidate nest failure model. Model-averaged parameter estimates (β) were calculated from all candidate models. Shading indicates inclusion of the variable in the model. ΔAICc is the difference in AICc from that of the best model. Model-averaged parameter estimates (β) are presented with confidence intervals using unconditional standard errors.

Model selection using AICc identified three best-fitting models of nest failure in the incubation and brooding stage, as two models fell within two AICc units of the most-supported model (Table 8.2.). However, breeding season and guarding received no support following model averaging, and distance to settlement was only weakly supported; removing distance to settlement and breeding season from the best model increased model AICc by 1.95 and 0.99 respectively. Model parameters indicated that DFR was greater with increasing distance to settlement (Fig. 8.2.). The best model predicted a 0.27 reduction in probability of overall success of nests located 10 km rather than 1 km from settlement in 2009–10, and a 0.54 reduction over this distance in 2010–11. Models including the distance to settlement term again found similar DFR among guarded and unguarded nests.

8.4.3. Conservation activity and nest failure

Nest failure was lowest at Kulen Promtep (Table 8.3.), appreciably lower than at Lomphat and the Mekong and marginally lower than at Western Siem Pang (including guarded and unguarded nests). The greatest contrast in nest failure was between Kulen Promtep, where only one (8.3%) of 12 nests failed, and Lomphat, where five (55.5%) of nine nests failed;

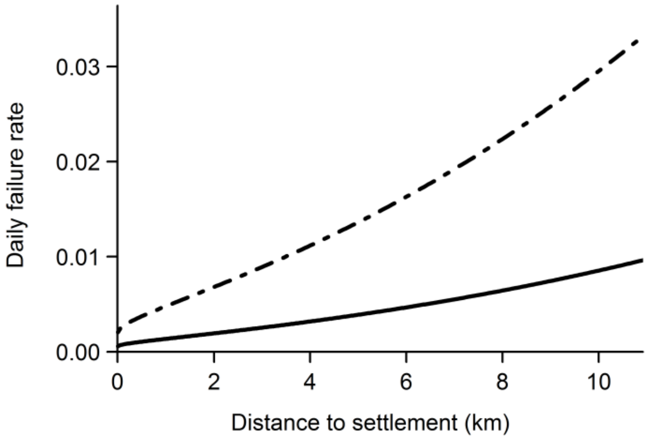

Figure 8.2. Daily failure rates (DFR) of nests during the incubation and brooding stage by distance to settlement and breeding season. Breeding season comprises 2009–10 (solid line) and 2010–11 (dashed line). DFR was predicted by the best-fitting binomial logistic regression model of nest failure at Western Siem Pang (model 6 in Table 8.2.; model AICc = 72.12, dispersion ratio = 1.07).

	β	95% CL	ΔAICc
Intercept	–6.29	2.03	
Study site:			1.87
Kulen Promtep	–	–	
Lomphat	2.36	2.19	
Mekong	2.22	2.10	
Western Siem Pang	1.74	2.02	
Breeding season:			–1.33
2008–09	–	–	
2009–10	–1.11	1.37	
2010–11	–0.19	0.86	

Table 8.3. Parameter estimates for a model of White-shouldered Ibis nest failure across the whole nesting period (incubation to fledging) including study site and breeding season. Kulen Promtep and 2008–09 were reference levels for study site and breeding season respectively. Breeding season is included to account for unequal sample sizes across seasons and study sites. ΔAICc is the change in model Akaike Information Criterion (AICc) when the term is removed from the model. Model AICc = 205.49, dispersion ratio = 1.41.

other pairwise comparisons between sites were not well supported. However, study site was not well supported in an overall model of ibis nest failure rate; removing this term caused a deterioration in model fit of less than 2 AICc units (Table 8.3.). Site-specific estimates of nest failure rate were significantly and negatively related to the ranking of sites by conservation activity ($r_s = –1.00$, $P = 0.042$; Fig. 8.3.), with greatest failure at the sites with most recent inception, the Mekong and Lomphat. However, this may not provide evidence that nest success is enhanced by conservation activity (relative to little or non-intervention), as five nests failed as a result of recently introduced conservation activity itself (jealousy-driven destruction of guarded nests at the Mekong), not because conservation was unsuccessful in alleviating other nest failure causes. Excluding these five nests, study site was supported as a predictor of ibis nest failure (model fit increased by 2.26 AICc when the term was removed) but there was no relation between nest failure rate and conservation activity ($r_s = –0.80$, $P = 0.167$).

8.5. Discussion

This study reports the first randomised experimental test of nest guarding, using White-

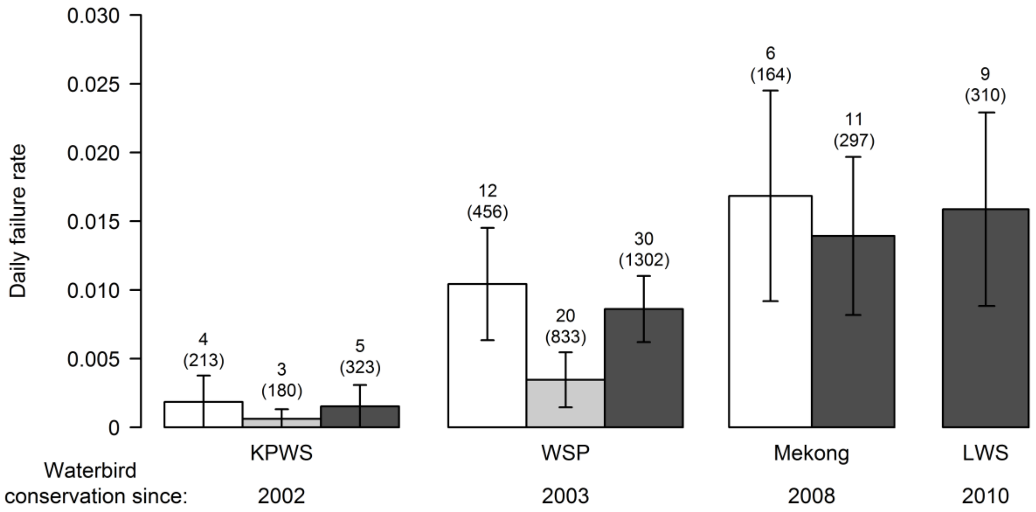

Figure 8.3. Daily failure rate (DFR) estimates of White-shouldered Ibis nests by study site and breeding season in order of conservation activity duration. Nests were monitored over three breeding seasons: 2008–09 (white); 2009–10 (pale grey); and 2010–11 (dark grey), although data were not available for every breeding season in Lomphat and at the Mekong. DFR was estimated using a binomial logistic regression model (Table 8.3.). The number of nests and the number of exposure days (parentheses) are given above each column; error bars indicate standard errors. The year that waterbird conservation began at each site is given beneath. KPWS = Kulen Promtep Wildlife Sanctuary, WSP = Western Siem Pang, Mekong = Mekong Flooded Forest, LWS = Lomphat Wildlife Sanctuary.

shouldered Ibis as a model species. The intervention proved ineffective for this species in Western Siem Pang, perhaps reflecting a greater impact of natural predation than of anthropogenic interference on nests at this site. Nest guarding effectiveness is likely to be context-specific and may vary between opportunistically exploited species such as White-shouldered Ibis and other species targeted for trade. Guard payments that accrue to only a few individuals can be a source of local discontent; implementing nest guarding schemes may therefore require caution, particularly at sites where intervention is recent and local conservation awareness is low.

8.5.1. Causes of nest failure

White-shouldered Ibis nest failures were caused by human exploitation, natural predation and high winds, problems that also affect nesting Giant Ibis *Thaumatibis gigantea* (Keo et al. 2009) and Lesser and Greater Adjutants, *Leptoptilos javanicus* and *L. dubius* in Cambodia (Clements et al. in press; Sok et al. 2012). Quantifying the relative importance of these causes of failure is not possible, as predation and human disturbance may have been disproportionately undetected. Furthermore, reward schemes could have ameliorated human impacts by providing an incentive not to disturb nests; this was most likely at Western Siem Pang and Kulen Promtep where schemes were applied for longest and with good staff capacity, creating relatively high local awareness. While failures were anthropogenic in more than a quarter of cases (64% of known-cause failures), more than half of these were provoked by resentment towards conservation intervention at the Mekong, perhaps relating to poor local awareness combined with a lack of guard diligence (Sok et al. 2012).

Natural predation caused failure of at least one nest and brood reduction at another, but may have caused other, undetected failures also. Fewer nest failures occurred in the late-nestling stage, perhaps because the chicks were too large to be predated or too advanced to be abandoned by disturbance-wary parents. Given, however, that humans are more likely to exploit nest contents at the late-nestling stage than at any other period (owing to greater conspicuousness in the nest and greater food value of chicks), higher failure during incubation and brooding suggests natural predation may be a more prevalent cause of nest failure at Western Siem Pang. Further research should assess the sources and levels of natural predation on ibis nests and the impact of human disturbance – particularly flushing adult ibis from nests – on their susceptibility to predation. As more nests are located across all sites, monitoring will help to quantify the relative contributions of natural predation and human interference to ibis nest failure, particularly if remote surveillance systems, such as miniature digital nest cameras, can be applied (Bolton et al. 2007).

Natural predation may also explain the positive relationship between nest failure and distance to settlement. Predators such as civets and martens are likely to be more abundant in remote parts of forest owing to strong hunting pressure, largely for trade (Srikosamatara et al. 1992). The estimated 67% decline in overall nest success from one to 10 km from settlement (2010–11) may relate to greater mammalian predation at remote nests; such predation also occurs at Giant Ibis nests in scarcely populated Cambodian dry forests (Keo et al. 2009). By controlling these predators, it is possible that humans may have indirectly protected nests close to villages; nevertheless, these conclusions are provisional as distance to settlement was only weakly supported in models of nest failure.

8.5.2. Nest guarding effectiveness

The study found little evidence that nest guarding was effective at Western Siem Pang, as daily failure rates did not differ between guarded and unguarded nests. While it is not possible to unequivocally conclude a null effect of guarding, the failure to improve nest success by at least a third (the minimum detectable effect size) calls into question the cost-effectiveness of the intervention for this species, at this site. Guard salaries were equivalent to US$246 per nest, indicating the substantial finance required if nest guarding were to be applied to a large proportion of the dispersed breeding population. The null effect is unlikely to be a result of ineffective protection, as guards (present during all daylight hours) were regularly checked on unannounced visits and were seen to intercept passers-by successfully, suggesting that they would have prevented actual cases of human interference. Rather, the result provides another indication that natural predation, not human exploitation, may be the greater threat at Western Siem Pang; attaching plastic baffles to nest trees, deterring mammalian predators (Keo et al. 2009), could be a valuable alternative to guarding here.

This study’s results differ from those of Clements et al. (in press), who found that guarded Lesser Adjutant and Sarus Crane *Grus antigone* nests had substantially higher success rates than unguarded nests. While adjutant and crane nests are routinely targeted for trade, White-shouldered Ibis nests are exploited only opportunistically and for consumption (Sok et al. 2012; HLW pers. obs.); contrasting results may therefore relate to different magnitudes of exploitation threat, with nest guarding effective at nests of traded species but having little impact at nests of lower-value species. However, Clements et al. (in press) contrasted guarded nests in protected areas with unguarded nests in unprotected areas, so that the apparent positive effect of nest guarding may, in part, also reflect changes to local attitude and behaviours brought about by other conservation interventions, such as community-based ecotourism, an agri-environmental scheme and law enforcement.

Empirical tests of conservation interventions face numerous methodological challenges (Ferraro & Pattanayak 2006). Testing nest guarding at a single site in this study enabled an assessment of its independent effect, but local awareness of the intervention could have potentially discouraged exploitation at *all* nests, guarded or unguarded. While this “spillover effect” (Pattanayak et al. 2010) cannot be ruled out, DFR did not differ between seasons during and prior to the nest guarding experiment, suggesting that nest survival was not uniformly improved in this way. An alternative method is to apply intervention and control treatments at separate sites, using statistical approaches to control for confounding factors (Ferraro & Pattanayak 2006); however, other conservation measures may conflate with the tested intervention if they occur concurrently and not in controls. In reality, it can be difficult to identify study communities unaffected by some form of conservation action or sites not conflated by other activities. The presence of a research team for several years may itself contribute to local awareness and change local behaviours; analysts must be wary of such constraints when evaluating interventions.

8.5.3. Conservation activity and nest success

At the larger scale of this study, four sites (among which nests showed a similar mean distance to settlements) were compared to assess the impact of conservation activity, as a whole, for White-shouldered Ibis nest success. Failure rates differed between two sites and were negatively correlated across all four sites with the rank of conservation duration (a proxy for the degree of activity); overall nest success was 63% lower at the site with least conservation activity compared to the site with most. However this trend must be treated with caution, as overall differences among sites were weakly supported. Furthermore, this trend reflects nest failures caused by conservation activity itself, rather than conservation alleviating an underlying cause of failure; the correlation was non-significant when these cases were excluded. This study therefore provides no evidence that conservation activity enhanced nest success, although detecting any such effect will have been limited by small sample sizes and potentially confounding differences in ecological and social conditions across the sites. The result provides some evidence for a potential detrimental effect of guarding at sites with little previous conservation history, as the jealous destruction of nests reflects local discontent with the distribution of payments. Nest guarding programmes that reward only a small proportion of the local community require careful implementation, as distributive unfairness has the potential to undermine the success of payment schemes (Sommerville et al. 2010). Improving community engagement measures and guard payment structures (e.g. payments conditional on nest outcomes) may address perceptions of unfairness and protect nests from inadvertently increased destruction (Sok et al. 2012).

8.5.4. Nest guarding: a useful tool for exploited species?

With guarding found to be ineffective in this study but effective in another (Clements et al. in press), the value of this intervention may be context-specific. Although the prevalence of anthropogenic nest failure could not be readily assessed, it remains likely that opportunistic nest exploitation, such as that for White-shouldered Ibis, occurs most frequently at sites with poor local conservation awareness. Further experimental tests may find that nest guarding is worthwhile in these circumstances, so long as local disquiet over guard payments is given careful attention. Nevertheless, nest guarding may be most valuable for routinely-targeted waterbirds – with a higher trade value than the ibis – that face a greater threat from next exploitation. Conservation programmes should continue to monitor the effect of nest guarding schemes, applying a control treatment of unprotected nests wherever possible, allowing for more comprehensive evaluation of this intervention's effectiveness.

8.6. **Appendix**: Nest stage durations and number of fledglings

From the sample of 100 White-shouldered Ibis nests, an average of 1.8 ± 0.6 (mean ± SD) chicks fledged per successful nest, with 21 March (± 24.9 days SD) the average fledging date across all years (the mid/late dry season). Three chicks were raised at six nests; these fledged an average of 21 days earlier than nests with one or two chicks, although this difference was not significant. The incubation and brooding stage averaged 43.8 ± 2.5 days combined (mean ± SD, $n = 17$), with the late nestling stage lasting 25.3 ± 6.4 days ($n = 27$) and the overall nesting period taking 67.6 ± 5.9 days ($n = 20$). Incubation took an average of 30.4 ± 2.7 days ($n = 17$), but this estimate is provisional as laying and hatching dates were hardest to determine accurately using ground-based observations. Ibis nesting duration (days from incubation to fledging) was shorter than for other dry forest waterbirds, such as Giant Ibis (Keo et al. 2009) and Lesser and Greater Adjutants in Cambodia (Clements et al. in press; Sok et al. 2012).

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[1] In addition to the papers presented in this thesis, the author also contributed to this paper, discussing the lessons learned from nest guarding in the Mekong Flooded Forest. The author provided comments on the content and structure of the manuscript and assisted with editing the text.

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Chapter 9

Conclusions

Via a new main road, the people of Siem Pang district have, for the first time, year-round road access to the nearest market town and beyond.

9.1. Key research findings

Agriculture and conservation are commonly seen as incompatible in the developing-world. Where the conservation value of farming is considered, it is often interpreted only in terms of its supporting role to biota in remaining natural habitats. This thesis took a different perspective, studying the value of agriculture to biodiversity now reliant on low-impact, traditional farming practices. The conservation challenges raised by these synanthropic species were explored in a detailed case study of White-shouldered Ibis *Pseudibis davisoni*.

The first chapter of the thesis revealed that a non-trivial number of developing-world birds are largely or entirely dependent on low-impact farming. Examples came from all regions of the developing-world and a broad range of, often traditional, farming systems. Crucially, these synanthropic, open-habitat taxa now lack natural habitats and have therefore become reliant on agriculture for their survival. They represent a distinctive but threatened subset of species and demonstrate that, in certain circumstances, agriculture not only retains biodiversity but in fact holds unique conservation value. Imminent agricultural transition necessitates greater attention to these farmed landscapes, where conservation may be forced to adopt mechanisms that maintain or mimic valuable farming practices.

White-shouldered Ibis foraging ecology exemplified synanthropic mechanisms in a traditionally farmed landscape. Ibis showed multiple associations with farming practices, as habitat use incorporated a range of anthropogenic and semi-natural habitats. Breeding ibis almost exclusively fed at *trapaengs* (waterholes), where vegetation was significantly reduced by extensively-reared livestock. In the wet season, ibis used *veals* (grasslands) and abandoned paddies created by low-intensity cultivation, most likely benefiting from access to the ground in these artificial clearings. Deciduous dipterocarp forest was also used, where grazing and fire – both components of the extensive livestocking system – reduced understorey vegetation, causing a predicted increase in ibis incidence. Finally, foraging White-shouldered Ibis occurred in closer proximity to people than Giant Ibis *Thaumatibis gigantea*, reflecting either a greater tolerance of disturbance, selection of anthropogenic habitat (abandoned paddy) and/or a stronger reliance on land management practices.

The case of White-shouldered Ibis also illustrated the challenge of reconciling conservation and livelihood change in low-impact farming systems. Local households derived considerable income from dry forest resources and the farming practices of benefit to ibis: rice cultivation provided the second greatest income of any activity and livestock provided an important capital asset – its value often exceeding total household income. Despite the likely reliance of both people and ibis on traditional farming, other evidence erodes the basis for a win-win conservation strategy: agricultural modernisation, shown by increasing tractor purchases, may be improving livelihoods but undermines the livestocking system of benefit to the ibis. Where such socio-economic changes threaten their loss, conservation will require new mechanisms to sustain valuable farming practices.

9.2. Conservation in developing-world agriculture

9.2.1. Patterns of synanthropy

The thesis presents initial evidence of synanthropy in farming landscapes of the developing world. Twenty-nine bird species were found to depend on agriculture, but the true number (and variety of taxa) may be much greater than available evidence can indicate: few studies have explicitly considered the role of humans in maintaining open and semi-natural habitats, or the biodiversity value inherent to developing-world agriculture. With data lacking, it is difficult to detect trends in these synanthropic relationships, but preliminary observations can be made. Most examples were found in Africa and Asia, perhaps due to the naturally widespread occurrence of open habitat (Woodward et al. 2004), the long history of agriculture (Mazoyer & Radar 2006), and/or the similarity of domestic grazers (particularly bovids) to their wild relatives in these regions; the paucity of examples from South America is intriguing and deserves further study. Mixed-farming systems may have particular significance, providing a heterogeneous mix of habitats and resources (van der Weijden et al. 2010), but pastoral systems also appear important, relating to the valuable role of grazing – now provided by livestock – for certain ground-dwelling avifauna. Species such as White-shouldered Ibis may benefit from both of these mechanisms.

9.2.2. Extending the European paradigm?

The thesis findings show wider relevance for a European paradigm: that of maintaining intermediate farming levels for conservation of biodiversity in anthropogenic or semi-natural habitats (Sutherland 2004). Where open-habitat taxa are dependent on developing-world farming, habitat management techniques (e.g. Sutherland & Hill 1995) and scientific knowledge of farming impacts (e.g. Robinson & Sutherland 2002; Tscharntke et al. 2005) could be transferred from Europe and the developed world. Research should evaluate the transferability of European techniques and knowledge by comparing valuable farming practices and management regimes between regions. European policies, such as agri-environmental schemes, have shown mixed success (Kleijn et al. 2006; Kleijn & Sutherland 2003), but when adequately targeted and evidence-based (Batáry et al. 2011) they too could be useful elsewhere. Nevertheless, obvious ecological, social and political differences may limit the wider applicability of the European experience, and local knowledge will also be vital. Above all, enhancing conservation research in the developing world, to even partly match European knowledge of farming systems, would be a worthy goal.

9.2.3. Informing the land-sharing versus land-sparing debate

Conservation science is seeking strategies to reconcile biodiversity conservation with increased agricultural production (Balmford et al. 2012; Green et al. 2005). Open-habitat taxa that lack natural habitat will be best conserved by a land-sharing approach, maintaining or adopting the wildlife-friendly farming practices that they now depend on. Agricultural intensification, proposed to spare land for nature, would likely assist the extinction of these species. Farming-dependent species will, of course, only comprise one subset of biodiversity in a given landscape and land-sparing may conserve other taxa more efficiently. Determining the best strategy requires detailed assessment of species' relationships with agriculture; methods examining population densities relative to yield (Green et al. 2005; Phalan et al. 2011) provide useful tools and deserve application in a range of contrasting landscapes.

Land-sparing and land-sharing are often proposed as opposing solutions (e.g. Phalan et al. 2011), creating a somewhat unhelpful dichotomy. In circumstances where habitat is uniform it may be possible to apply a single strategy; for example land-sparing in frontier ecosystems dominated by forest. However, in complex landscapes containing a mix of closed, natural and open, semi-natural habitat, species will exhibit contrasting tolerances and/or dependencies with agriculture, and simplifying conservation need will be less straightforward. Adopting a single strategy that conserves the largest proportion of a region's species will be dangerous if the remainder are lost as a result. A better solution may be to integrate the two strategies (Fischer et al. 2008; Norris 2008), allowing a broader range of species to be conserved in heterogeneous landscapes and perhaps also suiting the varied patterns of land ownership in the developing world (Adams 2012).

9.3. White-shouldered Ibis conservation

9.3.1. A dependency on traditional mixed farming?

Historically, wild herbivores such as Wild Water Buffalo *Bubalus arnee*, Gaur *Bos gaurus*, Banteng *B. javanicus*, Kouprey *B. sauveli* and Asian Elephant *Elephas maximus* may have been important ecosystem engineers (Timmins 2008). Grazing and wallowing at *trapaengs* may have sustained open habitat, providing an important ecological service to White-shouldered and Giant Ibises that require exposed substrates. These grazers may even have maintained or created these wetlands by preventing succession and removing sediment. The twentieth-century loss of natural herbivores (CEPF 2007; Loucks et al. 2009) may have added considerable importance to the role of traditional farming practices in keeping dipterocarp forests, *veals* and *trapaengs* open, with domestic livestock now mimicking the ecosystem functions once fulfilled by their wild cousins.

With the majority of foraging habitats now shaped or created by local farming, the persistence of White-shouldered Ibis populations is likely to depend on traditional agriculture. Although a positive effect of livestock grazing on ibis incidence and dry-season foraging success at *trapaengs* could not be shown (concealed in this study by ubiquitously short vegetation conditions), it is nonetheless likely to be vital. The ibis is known to feed in places with short or absent vegetation (Wright et al. 2010) and its small body size may limit its access to habitats with tall, dense vegetation stands (such as those observed in the wet season or at other sites). A study landscape with a steeper livestock density gradient would be valuable to assess the relationship between grazing intensity and habitat availability, and knowledge of optimal livestock densities, and burning regimes, would be particularly informative to conservation.

9.3.2. Valuable livelihoods with an uncertain future

Following this thesis’s findings, conservation in Cambodia has begun considering ways to support valuable, dry forest livelihood practices. At the time of writing, a pilot programme is testing whether the provision of free livestock vaccination encourages local people to keep their herds, thereby maintaining grazing of dry forest habitats. Recognising the value of farming is an important step for ibis conservation, with implications not only for habitat management but also for site designation. Traditional farmlands supporting ibis, as well as three severely threatened vultures species (Clements et al. in press-a) and Bengal Florican

Houbaropsis bengalensis (Gray et al. 2009), deserve protection alongside the more natural habitats that dominate Cambodia's protected area system.

Nevertheless, even where traditional farmlands are secured, livelihood transition is likely to become a major threat to White-shouldered Ibis. Evidence from Western Siem Pang and from northeast Thailand (Simaraks et al. 2003; Vityakon et al. 2004) suggests that livelihood development is not only imminent but may be profound, including the loss of free-grazing land and a concomitant decline in domestic buffalo numbers. Conservation interventions that address just one component of local livelihoods (such as livestock health) may be overwhelmed by the scale of this modernisation process. Incentives will need to be more sizeable and comprehensive if traditional grazing and rice cultivation practices are to be sustained in local communities. Alternatively, conservationists may be forced to take responsibility for managing dry forest landscapes themselves, for example, maintaining adequate grazing levels by purchasing livestock herds specifically for habitat management (at least until wild herbivore populations can be restored).

9.3.3. Further conservation considerations

Maintaining traditional farming practices is only one of several issues for White-shouldered Ibis conservation to address. While the vast majority of the ibis's global population is confined to Cambodia, 75% may occur outside of the country's protected areas. These populations need safeguarding from habitat loss as dry forests are under major threat of conversion to agricultural plantation, infrastructure and settlement (Clements et al. in press-b). Conservation should also take place at the landscape scale, as the ibis was found to require a variety of habitat types (which vary with season) and pairs dispersed across the dry forest to breed – probably reflecting a scarcity of prey at their favoured *trapaengs*. Although there was no evidence that the harvesting of amphibians by local people was currently impacting the ibis, an increase in harvesting (e.g. as other resources diminish) may lead to competition, with costs to ibis foraging during the breeding season. Human activity can also impact breeding White-shouldered Ibis, but nest guarding – employing local people to protect nests – did not improve nest success at the study site. Nest guarding may cause additional harm where schemes benefit only a small proportion of local communities, or where local people are unaccustomed with conservation action. Wet season foraging ecology and the significance of nest predation still require study.

Many of the thesis findings are of value to White-shouldered Ibis conservation efforts in Cambodia. These results were presented and discussed at a workshop, hosted by the author and BirdLife International, in Phnom Penh, January 2012, and attended by staff from governmental and non-governmental organisations. The main legacy of this study is a White-shouldered Ibis Coordination Group, established since the workshop to encourage collaboration between conservation organisations and to continue coordinated activities such as ibis censuses at roost sites.

9.4. Conserving valuable farmed landscapes of the developing-world

The conservation value of traditionally farmed landscapes is threatened by multiple drivers of agricultural change. Land acquisitions for externally-sponsored, industrial-scale plantation agriculture are not unique to the White-shouldered Ibis's range, and affect many parts of the developing world (Cotula et al. 2011). Other more local change occurs as farmers gain access to new markets and technology. The former may reflect far-reaching economic and political motivations while the latter encompasses local hopes of development – both powerful agendas that conservation must reconcile with the protection of farmland biodiversity. Added to this, provisioning an increasingly large and affluent human population demands widespread agricultural modernisation (Horlings & Marsden 2011), which will no doubt exert further change on these valuable, yet often low-yielding, farming systems.

Consistent approaches for addressing agri-business expansion and engaging local stakeholders are not yet forthcoming in conservation, despite close attention to the impacts of industrial-scale, intensive agriculture (e.g. Sodhi et al. 2010; Tilman et al. 2001) and a 25-year debate on conservation's role in fostering rural development (Roe 2008). Empowering local communities to defend their land-use entitlements (Cotula & Mathieu 2008) is one possibility for slowing the advancing frontier of industrial agriculture, but even where successful, this win-win approach does not guarantee that local people will continue using wildlife-friendly practices. Sustaining increasingly uneconomic traditional agriculture while meeting the development needs of local farmers will create a challenging trade-off for conservation. Although precise mechanisms remain uncertain (Adams 2012), farmers could be incentivised to maintain wildlife-friendly modes of farming (e.g. through direct conservation payments) and, where necessary, compensated for opportunity costs.

Restoring ecosystems is a possible alternative to conservation in low-impact farming systems (Phalan et al. 2012). However, this may often be impractical in current contexts, especially where unabated threats drive ecosystems even further from historic conditions (Hobbs et al. 2009). In the meantime, safeguarding farming's synanthropic species requires that conservation maintains, mimics or substitutes the valuable low-impact farming practices they depend on.

9.5. Next steps

This study provides only a starting point for understanding the value of developing-world agriculture to open-habitat biodiversity. Much of the thesis has focused on the synanthropic survival of White-shouldered Ibis which, although a single-species case study, demonstrates a phenomenon that may be widely overlooked outside of Europe. The prevalence of agriculture-dependent species needs thorough assessment so that their importance, relative to other global conservation priorities, can be understood. This research should extend beyond birds to examine a range of taxa, including both threatened and non-threatened species. Nevertheless, gaining this knowledge requires that agricultural systems of the developing-world receive greater attention in conservation science.

As traditional, low-impact farming systems are likely to change dramatically in the coming decades, research is urgently required to identify and understand the agricultural landscapes and practices of value to developing-world biodiversity; conservation will need to apply this knowledge where valuable modes of farming are lost. Many landscapes, particularly those comprising semi-natural habitats, need considering through a broader lens that acknowledges the role of humans in managing and sustaining valuable ecosystems. Finally, the challenges for conserving agriculture-dependent species are multi-faceted and clear solutions are not yet apparent. By seeking to maintain traditional farming conservation may be at odds with the development interests of rural communities; interventions are needed that explicitly and fairly address the balance between the well-being of local people with the persistence of wildlife.

This thesis highlights that, far from being incompatible with conservation, some farmlands are in fact critical to open-habitat taxa and deserve closer attention. After the widespread loss of species with agricultural modernisation, lessons learnt in Europe should be considered more widely to prevent similar wildlife declines in farming systems across the developing world. Open-habitat species are only one component of global biodiversity, but as agriculture expands and intensifies, they may feel some of the strongest effects of the increasing human population.

9.6. References

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