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Income and wildlife hunting in the Anthropocene: Evidence from Cambodia

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Ben Lockwood (Head of the Department of Economics, University of Warwick) and Michael Ward (Head of the Department of Economics, Monash University)

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Income and wildlife hunting in the Anthropocene: Evidence from Cambodia

Sharar Kader [*]

Abstract

Wildlife hunting is one of the largest causes of biodiversity loss, yet its drivers are still poorly understood. This paper quantifies the relationship between income and wildlife hunting in Cambodia, a country at the forefront of the clash between economic development and biodiversity loss. We use two nationally representative datasets which, unusually, collect detailed data on both the consumption and sales of hunted wildlife to estimate the importance of income on wildlife hunting in rural areas. Using rainfall shocks in the beginning of the main agricultural production season and prices of other protein sources as sources of exogenous variation in household income, we show that income has a causal negative relationship with wildlife hunting in rural Cambodia. We use these estimates to explore the effectiveness of cash transfers as a policy that promotes both wildlife conservation and poverty alleviation by primarily reducing the value of hunted wildlife as a coping strategy.

Key words: Biodiversity loss; Hunting; Rainfall shocks; Cash transfers; Cambodia
JEL classifications: O13, Q56, Q57, Q58

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1 Introduction

Over the last 50 years, the abundance of mammal and bird species declined by 83% and 58% respectively (Benítez-López et al., 2017), while 41% of amphibian species are currently threatened with extinction (IUCN, 2022). The IUCN currently includes over 40,000 species in its Red List of Threatened Species out of approximately 140,000 assessed (IUCN, 2022). These are snapshots of what has been labelled as the sixth mass extinction (Barnosky et al., 2011), and part of a larger process of environmental degradation driven by Human action that has pushed the Earth System into the Anthropocene (Crutzen, 2002).

While the biodiversity crisis has, until now, received considerably less attention than global warming (Cardinale et al., 2012), such large losses directly reduce a range of ecosystem services including forest composition (Bennett & Robinson, 2000; Harrison et al., 2016), primary productivity and carbon storage (Cardinale et al., 2012), and further threaten pollination, and pest and disease control in agricultural areas (Bradshaw et al., 2009; Sodhi et al., 2007). The Dasgupta Review on the *Economics of Biodiversity* (Dasgupta, 2021) provides a comprehensive analysis of the scale of impact such degradation precipitates, from the stability of food production to the preservation and restoration of large carbon sinks which are critical in reversing the effects of carbon accumulation in the atmosphere (see also Caro et al., 2022).

Given the significant spatial overlap between biodiversity hotspots and high levels of poverty (Fisher & Christopher, 2007), analysing the empirical relation between environmental degradation and poverty alleviation is crucial to understanding the extent of trade-offs between the two objectives. Recent evaluations on the impact of two cash transfer schemes on deforestation, in Mexico (Alix-Garcia et al., 2013) and in Indonesia (Ferraro & Simorangkir, 2022), offer conflicting evidence on this relation. Poverty reduction, via cash transfers, led to increased conversion of land to agriculture in Mexico, however, reduced the extraction of resources in Indonesia. As a result, the context of poverty reduction, and in particular the nature of income generation activities, likely matters in determining both the direction and magnitude of the effect of poverty reduction programs on environmental degradation.

Although much attention has been devoted to understanding the drivers of land-use change, in particular the role of agricultural expansion (see Pendrill et al., 2022), far less is known about the behavioural choices underlying wildlife hunting, an equally important driver of biodiversity loss (Caro et al., 2022; Tilman et al., 2017). Despite the importance of wildlife hunting as a driver of biodiversity loss, a recent review shows that most of the questions raised

in the early 2000s regarding the drivers of wildlife hunting (Milner-Gulland et al., 2003) remain unanswered almost 20 years later (see Ingram et al., 2021), particularly with respect to the relation between hunting and income, as emphasised in Brashares et al. (2011).

Conservation policy has operated in this vacuum, with policy responses to the biodiversity emphasising the imposition of bans on hunting wildlife and expanding conservation areas, as proposed in the Aichi Biodiversity Targets (Convention on Biological Diversity, 2020) and the Half-Earth project (Wilson, 2016). However, the effectiveness of these solutions has long been questioned, as inadequate resources and weak law enforcement have limited progress towards conservation aims (see Ingram et al., 2021), while already severely defaunated conservation areas may render the expansion of some conservation areas meaningless (Benítez-López et al., 2019; Harrison et al., 2016; Sreekar et al., 2015). In addition to trade regulation having minimal effect on reducing subsistence hunting (Pinedo Vasquez et al., 2021), these policies also often place the burden of conservation on the poorest members of society (Bennett & Robinson, 2000). Removing vital sources of nutrition and income without any replacement may increase food insecurity and economic deprivation, which can then exacerbate biodiversity loss via agriculture driven land-use change (Booth et al., 2021; Ingram et al., 2021).

In this work, we quantify the role of low income in the decision to hunt in Cambodia, a developing country that is both rich in biodiversity and where rural poverty is widespread. Contrary to most early research on this relationship, which has relied on relatively small case studies on the importance of wildlife hunting, we use nationally representative, detailed, household consumption surveys that, unusually, collect data on the importance of wildlife hunting at a household level. Section 2 presents this data and allows an initial, descriptive, discussion on the patterns of wildlife hunting in Cambodia and its apparent relation with shocks that impact agricultural production and poverty.

In Section 3, we first use OLS to analyse hunting behaviour in the context of household livelihood choices and find a negative, but relatively small, relationship with income. However, given that wildlife hunting is part of the livelihood portfolio of rural households, this relationship is unlikely to be causal. We address this concern by using rainfall shocks at the start of the main agricultural season and prices of substitute protein sources for wildlife (fish and seafood) to exploit plausibly exogenous variation in income. We additionally control for total rainfall in the wet season to account for any impacts decreased rainfall may have on the availability of wildlife. Our instruments clearly satisfy the statistical requirements of relevance and exogeneity, and allow us to estimate a negative and relatively high effect of income on

wildlife hunting: a decrease of 100,000 riels (US$24) in household per capita consumption correspondingly increases the value of wildlife hunting by ~6.2%. These results are robust to alternative econometric specifications that account for the high number of households that do not engage in hunting. In Section 4, we use these estimates to explore the use of transfer payments to households experiencing rainfall shocks. First, as a payment indexed to rainfall shocks, similar to ideas suggested by Chantarat et al. (2011) and the wider literature on index insurance, and second, as a cash transfer equivalent to the poverty line, conceptually similar to the idea of a Conservation Basic Income suggested by de Lange et al. (2022). We conclude in Section 5 with a discussion of future research questions raised by our results.

2 Wildlife hunting, agricultural production, and poverty in Cambodia

2.1 Context

We focus on Cambodia, a country that is one of the most biodiverse countries in Southeast Asia (Daltry, 2008) and sits at the heart of the Indo-Burma biodiversity hotspot, one of the most vulnerable biodiversity hotspots in the world (Fisher & Christopher, 2007). Wildlife hunting in Cambodia has led to the declines of most large wild vertebrates and remains the primary threat for most of the 100 known endangered vertebrates in the country (CBD, 2022; Harrison et al., 2016; Ibbett et al., 2021). While conservation areas, split between 63 designated areas, cover approximately 40% of the area of the country (UNEP-WCMC, 2022), these are also often emblematic of conservation problems worldwide, with insufficient funding and weak law enforcement blamed for failures in halting wildlife hunting and biodiversity losses (Ibbett et al., 2021; Souter et al., 2016).

As in other tropical Southeast Asian countries, rice production drives the agricultural economy of rural Cambodia, where most of wildlife hunting is also concentrated. Almost 60% of all agricultural land (comprising 38% of the Cambodia's area) is used solely for rice cultivation (USDA, 2022), which contributes to 80% of Cambodia's agricultural production (Vimean Pheakdey et al., 2017). Crucially, rice is predominantly a wet season crop as Cambodia's farming system is mostly non-irrigated and, as such, highly dependent on seasonal rainfall (Tsujimoto et al., 2020, Vimean Pheakdey et al., 2017). Rainfall shocks at the start of wet season are likely to be especially problematic, as they affect primary tillage and seedling transplanting (Sujariya et al., 2020), while also increasing competition between weed and rice plants (Vimean Pheakdey et al., 2017).

2.2 Data

We use data from the 2014 and 2019-20 Cambodia Socio-Economic Survey (CSES), an income and expenditure survey that, unusually, collects data on consumption and sales of wildlife and is representative at both national, and urban and rural levels. As it would be expected from the *ex-ante* literature, which suggests that proximity to the availability of wildlife is an important determinant of wildlife hunting, the majority of hunting occurs in rural Cambodia. We find 92% of households that hunt, who are responsible for approximately 94% of the value of hunted wildlife, are rural households. For this reason, we will focus on these areas for the remaining

of this paper. In the analysis that follows, we aggregate sales and consumption values of hunted wildlife per household to represent the total value of hunted wildlife per household. While the value of hunted wildlife may not always perfectly correlate with the impact of hunting on biodiversity loss, it is the most accurate proxy for analysis. All values are expressed in 2014 prices using the Consumer Price Index (World Bank, 2022).

Table 1 allows us to make some preliminary conclusions about the importance of wildlife hunting in Cambodia. The first conclusion is that, for almost 90% of the rural households who hunt, the value of wildlife consumption is larger than that of sales. While subsistence and commercial hunting occurs on a continuum and noting that data on the primary motivation for hunting wildlife is unavailable, we interpret this as suggestive that hunting in rural Cambodia is mostly driven by subsistence concerns.

The second conclusion is of substantial year-to-year variability in terms of the importance of this activity as the share of households involved in wildlife hunting in 2019-20 is almost three times its value in 2014. Given that the share of households that sell wildlife is largely unchanged (~1% in 2019-20 versus ~0.75% in 2014), this increase mostly reflects changes in the relative importance of households that consume wildlife. Also relatively constant is the value of hunted wildlife per household among households that hunt, further supporting the interpretation of wildlife hunting as a subsistence activity.

Finally, this data allows us to conclude that wildlife hunting is quite important for households that hunt. In both years, wildlife hunting corresponds to ~15% of the value of meat consumed by those households and is approximately equivalent to between two weeks and one month of total consumption (in 2019-20 and 2014, respectively).

Table 1: The importance of wildlife hunting in rural areas

	2014	2019-20
Consumes wildlife (%)	2.95	8.98
	(0.17)	(0.29)
Sells wildlife (%)	0.74	1.07
	(0.86)	(0.10)
Hunts wildlife (%)	3.11	9.13
	(0.17)	(0.29)
Value of wildlife hunted for all households (1,000 riels/year)	5.92	18.23
	(60)	(165)
Value of wildlife hunted for hunters (1,000 riels/year)	191	200
	(282)	(510)
Per capita total consumption (1,000 riels/year) for non-hunters	31.71	61.93
	(20)	(40)
Per capita yearly total consumption (1,000 riels) for hunters	27.56	48.47
	(14)	(24)
Non-hunters below poverty line (%)	32.57	17.70
	(0.47)	(0.38)
Hunters below poverty line (%)	37.45	30.76
	(0.48)	(0.46)
Wildlife consumed as share of meat consumption value for hunters (%)	13.95	16.54
	(0.20)	(0.44)
Observations	8,333	6,092

Notes: Mean values per household, standard deviation in parentheses. Own analysis of CSES 2014 & 2019-20. Monetary values at 2014 constant prices (CPI adjusted using World Bank data). Exchange rate as of 1 January 2014: 1 USD = 4,083 Riels; as of 1 July 2019 (CPI adjusted): 1 USD = 3,615 Riels (Source: www.xe.com). Wildlife share as value of meat consumption calculated using data on consumption from household food diary.

In common with other surveys designed to measure poverty and wellbeing in developing countries, the CSES collects rich data on variables such as overall consumption, which is an accurate measure of wellbeing in developing countries with large informal employment (Deaton, 2018), as well as household demographic characteristics, assets, and access to public services, including social safety nets. This data allows us to analyse the differences between households that hunt and those that do not, as summarised in Table 2.

In both survey waves, households that hunt are more likely to be headed by males and to be less wealthy (as measured by the value of durable assets they own) but with otherwise similar demographics (as measured by the dependency ratio). They are also more involved in agricultural production, as measured by their participation in growing crops, particularly rice, and their ownership of land and other agricultural assets (such as livestock and ownership of a fish pond), while also much less likely to own a non-farming business. However, this specialisation is not reflected in higher productivity: in both years, but particularly in 2019-20

(when participation in wildlife hunting is higher), agricultural output (as measured by rice yield) is lower for households that hunt.

These differences are summarised into one main factor in the decision for households to hunt wildlife: the value of wildlife hunting in rural Cambodia is significantly and negatively correlated with per capita consumption. We hypothesise that lower consumption is perhaps being driven by production shocks to the most important agricultural crop (rice). Specifically, we explore the relation between agricultural production and rainfall shocks in the critical period of the production calendar which subsequently impact income and, likely, increase the need for subsistence hunting.

Table 2: Wildlife hunting, household characteristics and wellbeing in rural Cambodia

	2014 & 2019-20		
	Non-hunters	Hunters	Difference
Per capita consumption (1,000 riels/year)	44.00	41.82	-2.18*
	(33)	(24)	
Below poverty line (%)	26.52	32.88	6.36***
	(0.44)	(0.47)	
Has a low income card (%)	16.22	19.75	3.53***
	(0.37)	(0.40)	
Value of durable goods (1,000 riels)	6,014	4,663	-1,351***
	(14,028)	(6,629)	
Owns livestock (%)	68.39	81.84	13.45***
	(0.46)	(0.39)	
Owns a business (%)	25.53	18.90	-6.64***
	(0.44)	(0.39)	
Owns a pond (%)	2.51	3.31	0.80
	(0.22)	(0.32)	
Land owned (hectares)	1.32	2.12	0.80***
	(2.40)	(2.69)	
Produces crops (%)	65.34	77.91	12.57***
	(0.48)	(0.42)	
Produces rice (%)	57.13	68.10	10.96***
	(0.49)	(0.47)	
Rice yield (kg/ha)	2,868	2,103	-764***
	(3,702)	(1,749)	
Dependency ratio	1.14	1.16	0.02
	(0.93)	(0.88)	
Age of household head	47.96	43.41	-4.55***
	(14.05)	(13.21)	
Household head is male (%)	78.60	90.55	11.96***
	(0.41)	(0.29)	
Observations	13,610	815	

Notes: Mean values per household, standard deviation in parentheses. Own analysis of CSES 2014 & 2019-20. Monetary values at 2014 constant prices (CPI adjusted using World Bank data). Exchange rate as of 1 January 2014: 1 USD = 4,083 Riels (Source: www.xe.com).

One important characteristic of the CSES is that it includes information that allows us to place surveyed households in space at a relatively disaggregated level, down to the commune level (the lowest administrative level in Cambodia). This, in turn, allows us to link our household data with a rich set of environmental variables, also at the commune level, that capture measures of natural capital, which may determine access to wildlife, and weather conditions, which may impact agricultural production.

As proxies for access to wildlife, we use data on biodiversity in 2005, measured through the Biodiversity Intactness Index (BII) which quantifies terrestrial species abundance and composition (de Palma et al., 2021), and the area of forest cover in 2000, which quantifies the importance of primary forests (Hansen et al., 2013). Additionally, we have data on the area equipped with irrigation in 2005 (Siebert et al., 2013), and soil conditions, namely soil depth (Shangguan et al., 2017) and available water capacity (Global Soil Data Task Group, 2000), which allow us to control for rice production conditions (Haefele et al., 2014).

We measure weather using data from the Climatic Research Unit gridded Time Series (CRU TS) (Harris et al., 2020). The CRU TS weather data allows us to construct different measures of rainfall, including total rainfall in the months of May and June previous to the survey (i.e., the first two months of the previous wet season) for the reasons mentioned at the start of this section: the reliance of rice production on wet season rainfall in (mostly) non-irrigated Cambodia and the crucial importance of the beginning of wet season for production.

This last aspect is illustrated in Figures 1 and 2, which present the partial linear regression estimates (Lokshin, 2006; Yatchew, 1997) of the relation between rice yield and rainfall in past May and June when we control linearly for environmental variables that may affect rice production at the commune level (soil depth, available water capacity, area equipped with irrigation and year fixed effects). We find rice yields are negatively influenced by values of rainfall below 150mm in May and 200mm in June, and largely unresponsive to values above this point. Although this relation is not causal, we will use them as a first approximation in the definition of rainfall shocks in our empirical analysis.

In support of earlier hypothesis, we show that rainfall shocks likely impact hunting behaviour by decreasing income, as shown in Figure 3, where rainfall shocks in the early wet season are negatively correlated with per capita consumption, and in Figure 4, where rainfall shocks in this period are positively correlated with values of hunted wildlife.

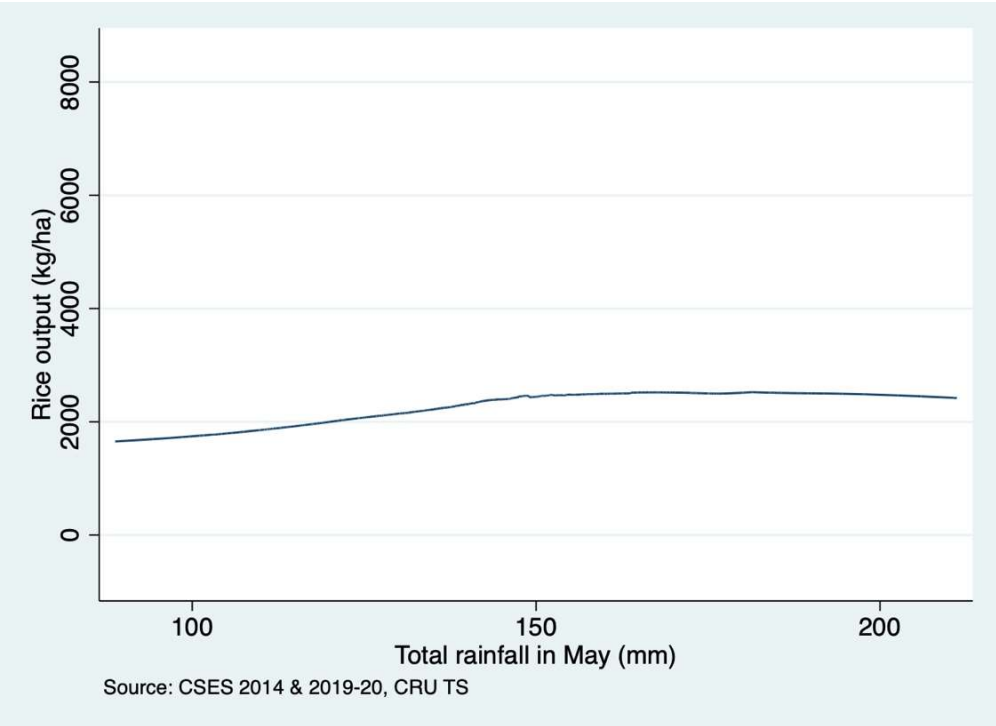

Figure 1: Rainfall in May and rice yield

Notes: N=8,090. Own analysis of CRU TS rainfall data and CSES household data, using values of rice yield from the previous wet season. P-value of total rainfall in May = 0.000, R-squared = 0.0092. 240 households considered outliers at the 5% significance level in the relation between rainfall in the wet season and rice yield are removed.

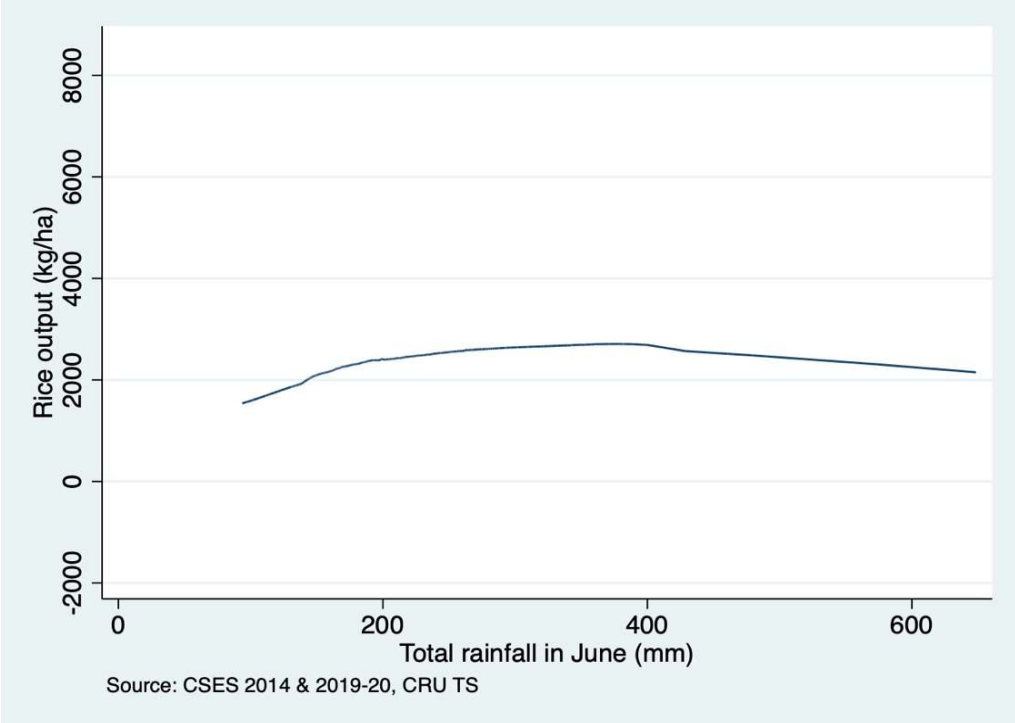

Figure 2: Rainfall in June and rice yield

Notes: N=8,090. Own analysis of CRU TS rainfall data and CSES household data, using values of rice yield from the previous wet season. P-value of total rainfall in June = 0.000, R-squared = 0.0082. 240 households considered outliers at the 5% significance level in the relation between rainfall in the wet season and rice yield are removed.

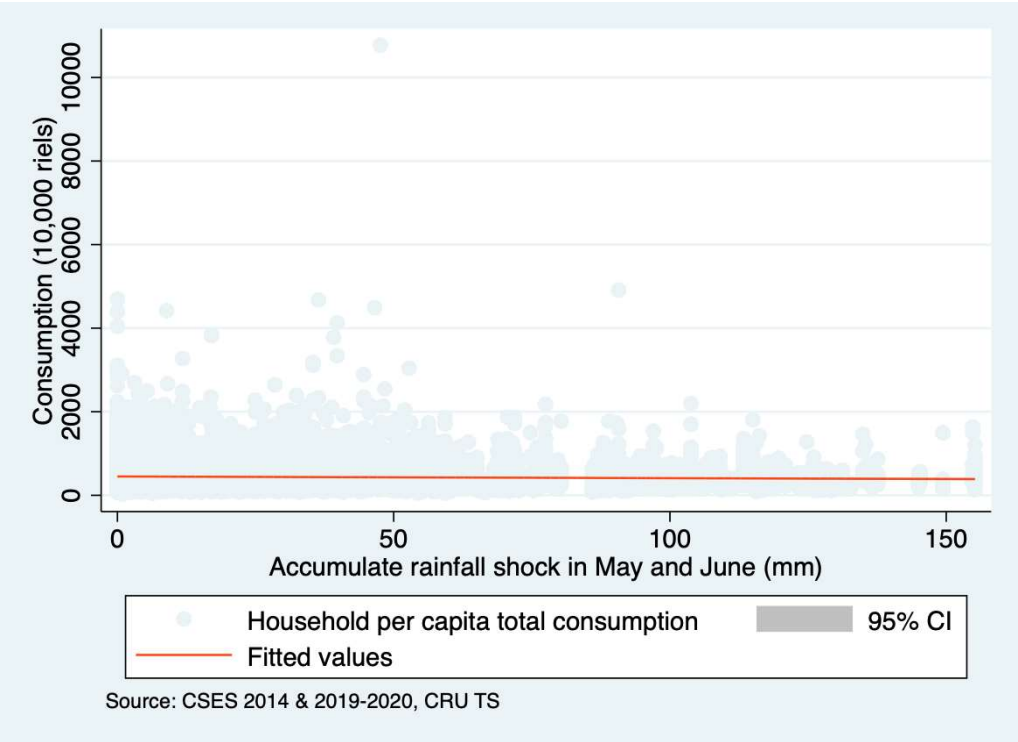

Figure 3: Rainfall in the early wet season and household consumption

Notes: N=14,425. Own analysis of CRU TS rainfall data and CSES household data, using annual values of consumption. Coefficient = -0.405, p-value of total rainfall in May and June = 0.000, R-squared = 0.0014.

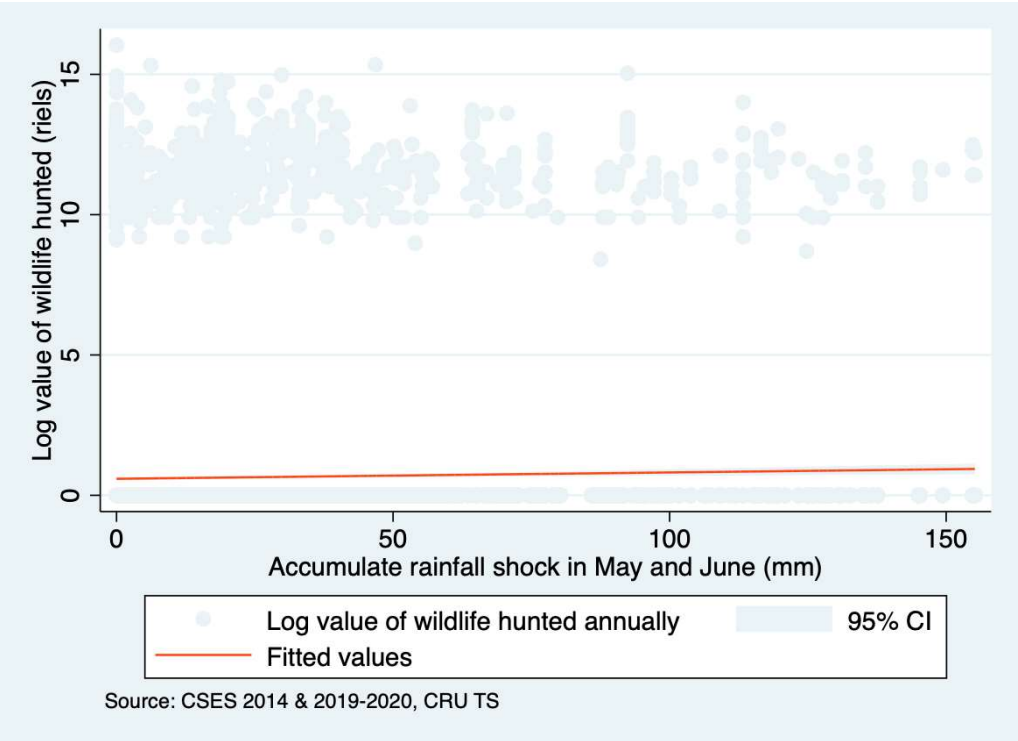

Figure 4: Rainfall shocks and wildlife hunting

Notes: N=14,425. Own analysis of CRU TS rainfall data and CSES household data, using annual values of wildlife hunting. P-value of accumulated rainfall shocks in May and June = 0.002, R-squared = 0.0007. Rainfall shock defined as the sum of |rainfall in May (mm) -150| and |rainfall in June (mm) -200|.

In Table 3 we complement our description of the patterns of wildlife hunting from Tables 1 and 2 with a spatial characterization of this activity which accounts for its local importance. We consider three types of rural communes: those where no surveyed households hunt (labelled as "no hunting"), those where wildlife is important (defined as those where more than 5 out of 12 households engage in wildlife hunting, labelled as "frequent hunting") and a residual category in between the two (labelled as "infrequent hunting"). Four main conclusions emerge from this Table.

The first is that communes where hunting is frequent (column 3) seem to be significantly different from communes where hunting is not reported (column 1) in terms of environmental conditions that could affect the availability of wildlife, including area of forest and biodiversity, which suggests that higher availability of wildlife lowers the opportunity cost of subsistence hunting.

However, and perhaps more importantly, our second conclusion is that they differ significantly in terms of the importance of rainfall shocks: across the first two months of the wet season in both years, communes where hunting is frequent experience significantly higher values of accumulated rainfall shocks than communes with no hunting. Given the much smaller importance of irrigation and lower soil depth, households in these communes are expected to be less capable to cope with such shocks. Likely as a result, rice yield and overall consumption in these communes are lower, and poverty rates are much higher. As suggested by the positive relation between rainfall shocks and wildlife hunting, the increased importance of wildlife hunting here seems to reflect the use of this activity as a coping strategy, used in response to agricultural production shocks.

Importantly, if shocks are driving low levels of income in these areas, the transitory poverty caused seems to be inadequately insured via existing social safety nets, as measured by being a beneficiary of the low income card. These transfer schemes are not correlated with the higher rates of poverty experienced in frequent hunting communes (compared to no hunting communes). This seems to suggest that existing safety nets are targeted to those who are persistently poor, while households who may fall in and out of poverty as a result of shocks typically remain uninsured.

Table 3: Wildlife hunting and local environmental conditions in rural Cambodia

		2014 & 2019-20	
	No hunting	Infrequent hunting	Frequent hunting
Per capita yearly total consumption (1,000 riels)	4,394	5,062***	3,844*
	(2,150)	(1,899)	(1,374)
Poverty rate (%)	26.60	23.61**	37.41***
	(0.21)	(0.18)	(0.24)
Has a low income card (%)	16.24	19.62***	17.69
	(0.17)	(0.18)	(0.17)
Value of wildlife hunted (riels/year)	-	28,093***	157,780***
	-	(48,693)	(192,823)
Yield from rice harvest (kg/ha)	3,041	2,510***	2,040***
	(2,182)	(1,703)	(909)
Median price of fish and seafood (riels/kg)	8,632	8,163**	6,855***
	(2,600)	(2,770)	(3,242)
Experiences a rainfall shock in May (%)	47.55	58.56***	73.47***
	(0.50)	(0.49)	(0.45)
Value of rainfall shock in May (mm), if shocked	7.41	11.87***	16.45***
	(13.55)	(15.66)	(15.09)
Experiences a rainfall shock in June (%)	73.78	61.60***	59.18**
	(0.43)	(0.49)	(0.50)
Value of rainfall shock in June (mm), if shocked	21.21	18.14	18.45
	(21.83)	(21.14)	(25.88)
Accumulated rainfall shocks in May and June (mm)	28.62	30.01	34.90**
	(29.55)	(31.51)	(35.75)
Total rainfall in previous wet season (mm)	1,419.16	1,446.25	1,570.68***
	(314.32)	(367.23)	(345.86)
Borders Vietnam (%)	56.50	33.46***	40.82**
	(0.50)	(0.47)	(0.50)
Borders Thailand (%)	17.48	34.22***	32.65***
	(0.38)	(0.48)	(0.47)
Biodiversity Intactness Index	0.88	0.92***	0.95***
	(0.07)	(0.07)	(0.06)
Forest cover (km^2)	12.43	21.83***	37.33***
	(20.44)	(24.04)	(30.01)
Area equipped for irrigation (%)	8.32	4.62***	3.17***
	(10.53)	(7.17)	(6.23)
Soil depth (cm)	9,948	9,588***	8,766***
	(1,283)	(1,285)	(1,754)
Water capacity in soil (mm)	232.16	232.78	233.03
	(9.90)	(13.93)	(8.23)
Number of communes	961	263	49
Number of households	10,967	2,910	548

Notes: Mean values per commune with standard deviation in parentheses. Infrequent hunting defined as 1-4 hunting households per commune and frequent hunting defined as 5+ hunting households per commune. Exchange rate as of 1 January 2014: 1 USD = 4,083 Riels. Rainfall shock defined as the sum of |rainfall in May (mm) -150| and |rainfall in June (mm) -200|. ***, **, * indicate statistical difference with 'no hunting' communes at the 1%, 5% and 10% level respectively.

Figure 5 presents these relationships between poverty and wildlife hunting at a provincial level. In both years, but more so in 2019 when hunting is higher, there is significant spatial overlap between poverty and the value of hunted wildlife. Crucial to our concerns about biodiversity loss, the spatial overlap of biodiversity and poverty observed at a global scale are repeated in Cambodia. Many of the provinces with higher values of hunted wildlife also host large conservation areas containing many endemic species and some of the last intact forests in mainland Southeast Asia (Daltry, 2008; Estoque et al., 2019).

Panel A – Poverty rates **Panel B – Value of hunted wildlife**

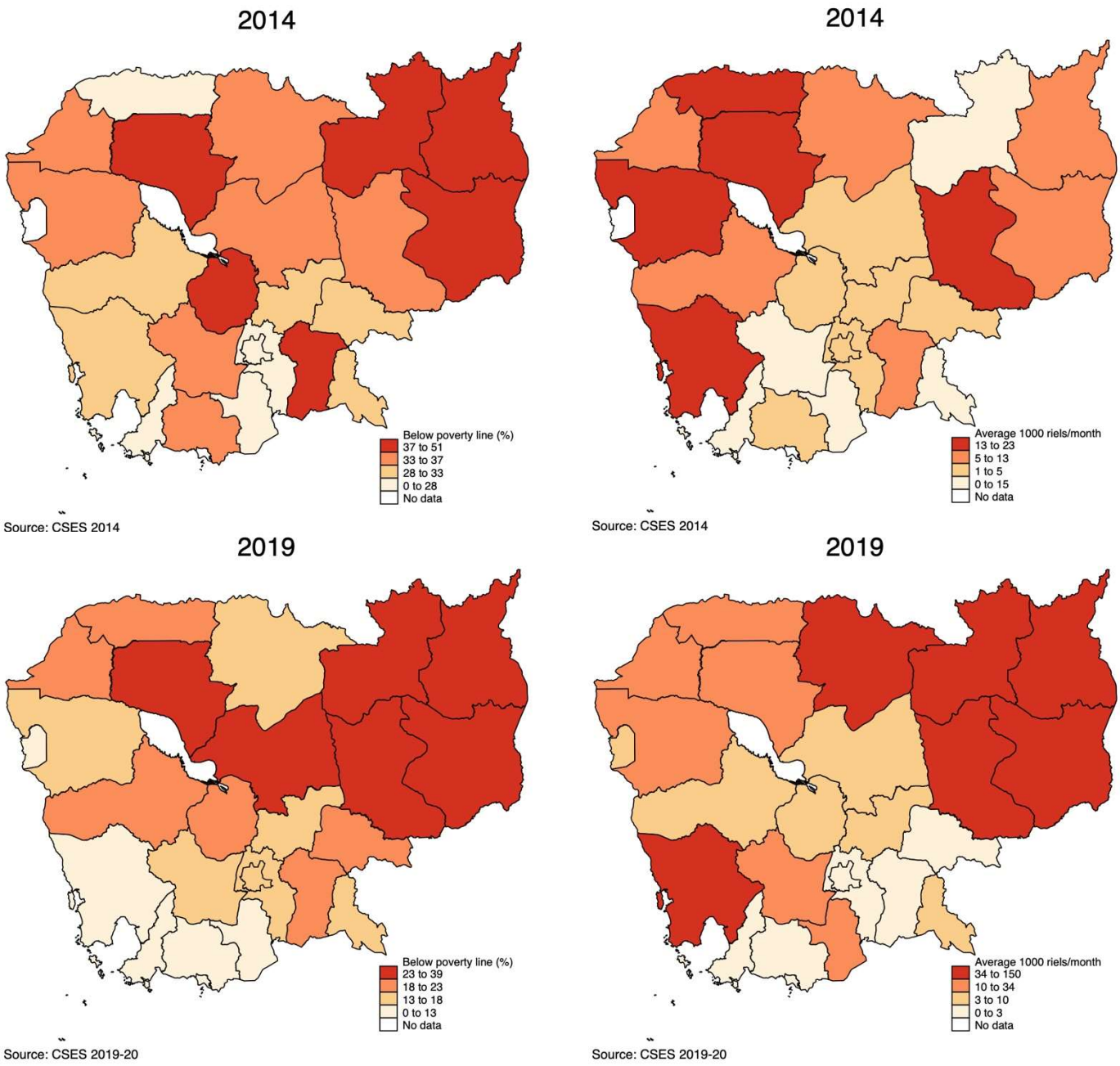

Figure 5: Poverty rates and value of hunted wildlife per household in rural Cambodia

Note: Own analysis of CSES data. Monetary values at 2014 constant prices (CPI adjusted using World Bank data). Exchange rate as of 1 January 2014: 1 USD = 4,083 Riels; as of 1 July 2019 (CPI adjusted): 1 USD = 3,615 Riels (Source: www.xe.com). In our 2014 figures, Tboung Khmum represents the same values as Kampong Cham as the CSES data collected in 2014 does not account for the provinces splitting in 2013.

3 Empirical estimates

The descriptive analysis presented in the previous section and the ex-ante literature suggest a negative relation between income and wildlife hunting in rural areas. In this section we formally test this hypothesis. We start by using OLS to estimate the following equation:

$$\log(W) = \beta_0 + \beta_1 Y + \beta_2 X + \lambda_3 D + \varepsilon \qquad (1)$$

where W represents the value of wildlife hunting in rural areas and Y is yearly per capita consumption, X are household control variables that matter for consumption, including demographic characteristics (such as age and sex of the household head) and wealth (such as ownership of livestock and non-agricultural businesses), and for wildlife availability, including environmental characteristics (such as total rainfall in the wet season and lagged measures of biodiversity and forest cover) and, finally, D stands for a year fixed effect for the survey year (2014 or 2019-20 CSES). Standard errors are clustered at the commune level, the level at which our environmental data is available, and our estimates are weighted using household weights provided by CSES. The OLS estimates are presented in Table 4, column (1).

Our primary conclusion is that, controlling for a large number of observed confounders of consumption and of participation in hunting, we find a negative relationship between income and wildlife hunting: a decrease of 100,000 riels/year in per capita consumption (~US$24, approximately 2% and 3.5% of total yearly per capita consumption in 2019-20 and 2014 respectively) increases the value of hunted wildlife per household by ~0.6%.

Although the estimates of other covariates are not the main focus of this analysis, our results seem intuitively reasonable.[1] Male headed households and those headed by younger heads, hunt much higher values of wildlife. Access to larger areas of forest, which we interpret as a proxy for wildlife abundance, leads to increased values of hunted wildlife, which are both large and precisely estimated. Finally, it seems clear that wildlife hunting should be seen as a part of the livelihood portfolio of rural households: household reliance on agricultural production, as reflected in land and livestock ownership, leads to a greater importance of hunting, which is attenuated by lower ownership of non-farming businesses. Better agricultural production conditions, as measured through soil depth and water retention capacity in the soil, decrease the importance of hunting, also suggesting that increased capacity to cope with rainfall shocks may decrease the importance of hunting. Importantly, being the beneficiary of a low

[1] Estimates of covariates are presented in Table A.1 in the Appendix

income card does not reduce hunting behaviour which suggests that the transitory poverty likely driving hunting behaviour is not adequately addressed by existing social safety nets.

One obvious limitation of these results, that may explain the low estimate of the effect of income, is that we do not account for the censoring of the distribution of wildlife hunting. To address this criticism, we estimate a Tobit model of this decision which is presented in Table 4, column (2) as a robustness check. Compared to our OLS estimates, the Tobit estimates are equally precise with the same direction and much larger magnitude. For all households, a decrease in per capita consumption increases the value of hunted wildlife per household by ~10.9%, a value that is almost twenty times the size of the OLS estimates presented in column (1). We also estimate the effect for households that are already hunting as policy responses may impact these households differently. Here, a decrease of 100,000 riels/year in per capita consumption increases the value of hunted wildlife per household by a notably higher ~14%[2].

The interpretation of wildlife hunting as part of a portfolio of income generation activities, including as a coping mechanism against shocks (de Merode et al., 2004; Duffy et al., 1993a; Enuoh & Bisong, 2014), suggests that our estimates of the effect of income on hunting are unlikely to have causal interpretations. It is also unlikely we would be able to adequately control for all variables that determine that same portfolio. For example, CSES does not collect data on risk and time preferences that are likely to determine those choices. To overcome this limitation, we exploit plausibly exogenous sources of variation in income by using rainfall shocks and the prices of substitute protein sources, in particular of fish and seafood as they are widely established to be direct substitutes of wildlife (Brashares et al., 2004; Nasi et al., 2008; Cawthorn & Hoffman, 2015; Duffy et al., 1993b)

To construct our measure of shocks, we explore our analysis of the relation between rice yield and rainfall at the start of the wet season. We use CRU TS weather data to define commune level shocks: the absolute value of the sum of rainfall below the thresholds at which increased rainfall stops influencing rice yield in May and June (150mm in May and 200mm in June, as shown in Figure 1 and 2). Despite its wide use in economics (see Dell et al., 2014 for a review), the use of rainfall shocks in our analysis is less straightforward than in many other contexts (see (Mellon, 2022) for a related discussion of the plausibility of rainfall meeting the exclusion restriction in many applications): we hypothesise that shocks impact income via agricultural production, while our dependent variable reflects a different form of biological

[2] Marginal effect estimated conditional on the value of wildlife hunting being greater than 0 (i.e., the estimate for households that hunt)

production (wildlife). For this reason, we include the total amount of rainfall during the wet season as an additional control. While wildlife is responsive to reductions in water availability, we assume that contrary to cultivated crops with a well-defined cropping calendar, this response is less dependent on the timing of rainfall.

In addition, we use median prices of fish and seafood at the commune level, which are CPI adjusted to price levels in 2014, using CSES data on the amount (kg) and price (riels) of these alternative protein sources. The use of the price of substitutes as instrumental variables relies on the assumption, with a long tradition in economic analysis (Angrist & Krueger, 2001), that individual households have no market power which seems reasonable in our context.

The Instrumental Variable estimates are presented in Table 4, column (4). Before we discuss these results, we note that the first stage regression, presented in column (3), creates sufficient exogenous variation in income by satisfying the weak instrument tests proposed by Stock & Yogo (2005), and satisfies the relevance requirements of this approach. Our two instruments also satisfy the overidentification constraints, further supporting our assumption that the exclusion restriction is satisfied in our case.

Our main conclusion is that a decrease of 100,000 riels in yearly per capita consumption increases the value of wildlife hunting by ~6.2%, a value that is much larger than the OLS estimates presented in column (1). Given the average rainfall shock faced by households (=29mm) and the associated income shock of ~136,000 riels/year (US$33, approximately 2.2% and 4.3% of per capita consumption in 2019-20 and 2014 respectively), we can conclude that the average shock increases the value of wildlife hunting by approximately 8.4%.

Similar to the discussion above, we address concerns about the censored nature of our dependent variable through the estimation of a Tobit IV model (Table 4, column 6). Our conclusions are robust to this specification. The effect of income on wildlife hunting is again precisely estimated and negative, however, their magnitude is much larger than the results presented in columns (3)-(4). According to the first stage estimates, the average rainfall shock leads to a decrease in income per capita of ~168,000 riels/year ($41) which increases the value of wildlife hunting by ~162,000 riels/year. For households already participating in hunting, who are likely to be of lower income, the average rainfall shock increases the value of hunting by ~12.9%[3], a result slightly slower than the Tobit estimates discussed for these households.

[3] Marginal effect for estimated conditional on the value of wildlife hunting being greater than 0 (i.e., the estimate for households that hunt)

In the next section we use the linear estimates presented in columns (3) and (4) of Table 4, given their ease of interpretation, to discuss the implications of our results for policy.

Table 4: Estimates of income on wildlife hunting in rural areas

VARIABLES	(1) OLS	(2) Tobit	(3) IV – First stage	(4) IV	(5) Tobit (IV) – First stage	(6) Tobit (IV)
Per capita yearly total consumption (10,000 riels)	-0.006***	-0.109***		-0.062***		-0.964***
	(0.001)	(0.017)		(0.017)		(0.126)
Household controls	Y	Y	Y	Y	Y	Y
Commune controls	Y	Y	Y	Y	Y	Y
Year fixed effect (2019-20)	Y	Y	Y	Y	Y	Y
Rainfall shock (mm)			-0.047***		-0.058***	
			(0.01)		(0.01)	
Price of fish and seafood (riels)			0.001***		0.001***	
			(0.00)		(0.00)	
Observations	14,425	14,425	14,425	14,425	14,425	14,425
R-squared	0.070					
Kleibergen-Paap rk Wald F statistic			23.692			
Hansen J stat			0.618			
Hansen J stat p-value			0.432			
IV joint-significance test = 0 (p-value)					0.000	
Correlation of error terms						1.001***
						(0.111)

Notes: Standard errors, clustered at commune level, in parentheses. Stock-Yogo weak identification test critical values for Kleibergen-Paap rk Wald F statistic reported for model (2): 10% maximal IV size =19.93. *** p<0.01, ** p<0.05, * p<0.1

4 Reconciling poverty alleviation with conservation: an ex ante analysis of two policy options

The role of wildlife hunting as a coping strategy in response to rainfall shocks, and what that tells us about the causal negative relation with poverty, allows us to propose alternative policy responses to the biodiversity crisis that do not rely on excluding the poorest from using natural resources. We use our estimates from the previous section to estimate the costs and impacts of transfer payments designed to alleviate the negative income effects of rainfall shocks in rural Cambodia and, consequently, reduce their incentive to hunt wildlife.

We discuss the potential use of two different options. In the first option, and similar to suggestions by Chantarat et al. (2011), we propose a Conservation Indexed Transfer (CIT) equal to the estimated income lost due to rainfall shocks at the start of the wet season (presented in Table 5, row 1). Based on our first stage estimates (Table 4, column 3), this payment corresponds to 4,657 riels per person per millimetre of rainfall below the critical levels for May or June which, given the average household size (4.5 people in rural Cambodia), translates to a payment of ~21,000 riels (US$5) per household per mm. Replacing the total income lost from rainfall shocks within our sample translates to $2.2 million in transfer payments, with an associated reduction of the value of hunted wildlife in rural Cambodia by 16%.

In the second option, we explore the idea of a Conservation Basic Income (CBI) to reduce biodiversity loss, as proposed by de Lange et al. (2022). However, contrary to the generalised transfers to all households in important biodiversity areas proposed by these authors, we target these transfers to households living in communes experiencing rainfall shocks. The CBI would correspond to a payment of ~2.2 million riels per person ($552, i.e., the yearly poverty line in 2014) which is equivalent to ~10.1 million riels ($2,484) per household, for all households in eligible communes. As this transfer is designed to eliminate poverty and not only the effect of the rainfall shock, the CBI would lead to much a larger reduction in wildlife hunting (62%). However, this option not only has a much higher total cost ($30.3 million), but is also much less cost-effective, with the cost of a 1% reduction in the value of hunted wildlife being almost four times the cost of the CIT.

We can contrast these payments with the total budget currently devoted to environmental conservation in Cambodia, via the National Environment Strategy and Action Plan 2016-2023 (NESAP) (Royal Government of Cambodia, 2017). This plan targets protecting the diverse environmental capital in Cambodia, including wildlife conservation, with a total budget of $263.5 million over the 7 year period. These two options targeting 12,215 households would represent approximately 0.8% and 11.5% of this budget.

The importance of these costs, particularly in the case of the CBI, raises the question of whether better targeted designs can be more cost-effective. The first, and perhaps most straightforward improvement, would be to target communes that only suffer large rainfall shocks, defined here as being above the median rainfall shock (28.5mm). This change results in nominally lower total benefits of wildlife conservation for both the CIT (of 12%) and CBI (of 31%), as presented in Table 5, row (2). The cost of a 1% reduction in the value of hunted wildlife is slightly higher than in row (1) for the CIT, which suggests that wildlife hunting in communes with large rainfall shocks is not solely driven by transitory reductions in income, as the value of cash transfers does not increase by targeting large rainfall shocks. The cost effectiveness of the CBI, which also targets the persistent aspects of poverty, does not increase as it is approximately constant. Transfers targeting the 25th and 75th quantiles of rainfall shocks[4] also do not increase the cost effectiveness for the CIT compared to row (1). However, the value of the CBI does increase when targeting the 75th quantile of rainfall shocks, which supports our interpretation that persistent levels of poverty also drive hunting behaviour as households are not experiencing poverty solely due to rainfall shocks. These results suggest that it may be worthwhile to target both types of payments in a more cost-effective manner than by simply relying on the importance of rainfall shocks.

An alternative approach is to direct these payments to households that are most likely to hunt wildlife. The feasibility of this approach relies on being able to select a few observable and non-manipulable characteristics that can predict the decision of engaging in hunting activities. We consider demographic characteristics of the household (sex, age and years of schooling of the household head, and household size), wealth indicators (ownership of a house and livestock), access to safety nets (low income card and pension) and commune characteristics (proximity to Thai and Vietnamese border, biodiversity, soil depth and water capacity) to estimate a logit model where the dependent variable is the decision to hunt[5]. We then use the predictions of this model to determine who is eligible for either of these cash transfers, by identifying households that are most likely to hunt.

First, we target 3,053 households who are in the top quartile of the probability of hunting in areas facing rainfall shocks (probability of hunting >6.7%). As shown in Table 5, row (3), CIT payments proportional to the value of rainfall shock faced by these households would have cost US$629,875 and reduced the total value of hunted wildlife by 11%

[4] Outcomes of targeting the 25th and 75th quartiles of rainfall shocks are shown in Table A.3, rows (2) and (4) in the Appendix

[5] Estimates of logit model on decision to hunt wildlife are presented in Table A.2 in the Appendix

(US$55,204 per 1% reduction in hunted wildlife). Limiting these transfers to the 610 households in the top 5% of likely hunters in these areas (probability of hunting >17.2%; Table 5, row 4) would have further increased the cost-effectiveness of the CIT (US$27,872 per 1% reduction in hunted wildlife); although the overall reduction in wildlife hunting is quite small (4%). These two scenarios would comprise of merely 0.24% and 0.04% of the current 7 year budget allocated to environmental protection.

Similarly, these changes in targeting also increase the value of the CBI when compared to rows (1) and (2). Targeting the top 25% of likely hunters in communities facing rainfall shocks would have reduced the value of hunted wildlife by a large amount (43% of the value without policy) but at a relatively high cost of $174,474 per 1% decrease. Further narrowing the targeted households to the top 5% of likely hunters still leads to a fairly high reduction in wildlife hunting (19% of the original value) at much lower associated costs ($80,516 per 1% reduction in value of hunted wildlife).

When contrasting the two approaches, it seems that targeting shocks alone (via the CIT) is likely to lead to relatively small reductions in wildlife hunting which may be undesirable from a conservation perspective. On the other hand, reducing poverty (via the CBI) is very effective in terms of reductions in the value of wildlife that would be spared (almost two-thirds of wildlife in row 1), however, it is likely too expensive to be justified on conservation grounds alone.

As a final analysis we combine targeting 298 households who are both in the top 5% of likely hunters and also experience above median rainfall shocks, as presented in Table 5, row (5). While the CIT is now less attractive, as its cost per 1% reduced value of hunted wildlife is slightly higher than in scenario (4), the CBI is significantly less expensive while still reducing wildlife hunting by 10%. This option of the CBI would comprise 0.28% of the current environmental budget.

We finalise our comparison of these two options by noting that, while the CBI is more expensive than the CIT as expected, solely focusing on the importance of shocks is unlikely to reduce wildlife hunting to sustainable levels. The CBI becomes increasingly more cost effective, relative to the CIT, as we narrow our targeting of the cash transfers, as shown in the final column of Table 5 (CBI:CIT). This reflects that larger transfers to households who are more likely to hunt wildlife, based on characteristics other than just rainfall shocks, increases the value of cash transfers. Thus, we conclude that overall poverty drives hunting behaviour, not just transitory losses in income from rainfall shocks, and therefore, a policy that mixes

instruments which target both the persistent and transitory nature of poverty is likely to be the most effective approach to biodiversity conservation.

Table 5: Cash transfers to reduce wildlife hunting

			Conservation indexed transfers (CIT) (~US$5/1mm)				Conservation Basic Income (CBI) (US$2,484/household)			CBI:CIT
Scenario:	**Target households**	**N**	**Average transfer (US$/hh)**	**Total cost (US$)**	**Change in hunting value**	**$/% hunting reduction**	**Total cost (US$)**	**Change in hunting value**	**$/% hunting reduction**	**$/% hunting reduction**
1	All households in communes facing rainfall shocks (>0mm)	12,215	$179	$2,186,544	-16.49%	$132,598	$30,342,060	-62.38%	$486,407	3.67
2	All households in communes facing above median rainfall shocks (>28.5mm)	6,121	$294	$1,798,055	-12.43%	$144,654	$15,204,564	-30.98%	$490,786	3.39
3	Top 25% of likely hunters | commune facing rainfall shock (>0mm)	3,053	$206	$629,875	-11.41%	$55,204	$7,586,136	-43.48%	$174,474	3.16
4	Top 5% of likely hunters | commune facing rainfall shock (>0mm)	610	$188	$115,112	-4.13%	$27,872	$1,517,724	-18.85%	$80,516	2.89
5	Top 5% of likely hunters | commune facing above median rainfall shock (>28.5mm)	298	$298	$89,037	-2.91%	$30,597	$725,972	-9.82%	$73,928	2.42

5 Conclusion and suggestions for further research

Wildlife hunting supports the livelihoods of millions of poor rural households in developing countries by providing vital sources of income and nutrition. Unfortunately, it is also largely practiced at rates that are considered unsustainable, making it one of the biggest causes of biodiversity loss. Reflecting our lack of knowledge about the drivers of wildlife hunting, policy responses to protect biodiversity have traditionally resorted to restricting access to resources while potentially overlooking trade-offs against outcomes for impacted populations.

In this paper, we examine the relation between poverty and wildlife hunting using a nationally representative survey from Cambodia which collected rare data on wildlife hunting at the household level. This allows us to investigate the role of income in driving wildlife hunting at a national scale and avoid the limitations of smaller case studies that have so far characterised this literature.

With wildlife hunting often employed as a coping strategy against negative income shocks, we instrument income using both rainfall shocks, which impact agricultural production and often trigger increased hunting among rural households, and prices of fish and seafood. We find that the reduction in income caused by the average rainfall shock leads to an associated increase in the value of hunted wildlife of ~8.4%. This finding carries important policy implications as insuring rural incomes from rainfall shocks can potentially reduce poverty while increasing biodiversity conservation, which are two objectives that have often been considered incompatible. By protecting households from falling into transitory poverty, which is largely not insured by existing social safety nets, insurance can prevent the need for households to hunt wildlife as a coping strategy.

We consider two types of transfers to households in communities affected by rainfall shocks. The first borrows from the large literature on index insurance and proposes Conservation Indexed Transfers (CIT) equivalent to the loss of income associated with the rainfall shock experienced at a local level. The second builds on recent discussions on the feasibility and interest of a Conservation Basic Income (CBI) which transfers a flat-rate payment set as a function of the poverty line. Although both offer insurance against agricultural risk, each approach addresses a different aspect of poverty. While the CIT aims at addressing transitory reductions in income from rainfall shocks (but offers no guarantee that households will not be poor), the CBI aims to reduce poverty directly.

A comparison of the relative size of the transfers (in US$ per household) and of the size of associated behavioural changes they induce (in US$ per 1% reduction in the value of hunted

wildlife) leads us to conclude that an optimal policy will likely include a combination of both approaches. However, there is no suggestion that the two approaches can yet be trialled and evaluated. We notice several important questions that need to be addressed regarding the scope of targeting these policies effectively and any potential negative spillovers that cash transfers may lead to.

The first, and most important, is on the assumption that such policies are valuable from the perspective of society. The true value of our estimated reductions in wildlife hunting likely depends on more than its monetary value alone, and should reflect, for example, the conservation status of the species that are spared. Although we have no information on the species being hunted in our data, we suggest that paying insurance for conservation is likely better targeted to regions rich in biodiversity. While we already control for such characteristics in some of the analysis, explicitly including areas surrounding national parks and wildlife conservation areas in the targeting criteria will likely increase the cost benefit ratio of both types of transfers. Noting that further research on the locations of hunting and the species being hunted is needed to target areas where risks to biodiversity loss are greater, it may be possible to proceed without such detailed information on the species composition of hunting behaviour.

The second difficulty in implementing either of the proposed transfer payments is the implicit assumption that data on local rainfall is available in a timely manner to allow for the planning and disbursement of these transfers. This is unlikely to hold true in many contexts and leads us to rely on what has been learned from the existing literature on index insurance (see Carter et al. (2017) for a review). As a suggestion, these transfers can be linked to measures of vegetation levels, which are readily available as spectral reflectance data from satellite sensors (using indexes such as the NDRE, NDVI, RVI and RERVI) and have also been shown to accurately predict yields in the early to mid-growth stages of agricultural production (Zhang et al., 2019).

Third, there remains a need for research into the seasonality of hunting behaviour to clarify whether it better reflects the seasonal nature of poverty, which increase towards the end of the wet season, or the opportunity costs of labour, which are lower during the dry season. This may matter if the effect of insurance on wildlife hunting is determined by the timing of the transfers. While our estimates are based on annual values of hunted wildlife, savings constraints suggest that transfers may be more effective if closer to when households consider hunting.

Fourthly, both our approaches are proposed as unconditional cash transfers. There is a limited, yet growing, use of payments for ecosystem services programs which condition cash

transfers on meeting criteria of sufficient provision of environmental protection. However, little of this focus has been targeted towards reducing wildlife hunting and, as with cash transfers more generally, whether conditionality matters remains an open question. In our context, enforcing conditionality of not hunting wildlife is difficult, if not impossible, without incurring high transaction costs. While we propose cash transfer programs based on the extrinsic motivations of wildlife hunting (i.e., income), and noting that some progress has been made into the context dependent cultural importance of wildlife hunting, the understanding behind the intrinsic motivations of hunting remains relatively limited. Particularly, the value that people place on hunted wildlife may be driven by more than its monetary and nutritional benefits, for example, and the intrinsic value of the activity of hunting itself may hinder the benefits of cash transfers.

Our last concern is regarding the indirect effects of providing insurance against rainfall shocks, either as indexed or flat-rate transfers. While reducing agricultural risk for households reduces biodiversity loss via wildlife hunting, it also increases agricultural profitability and, consequently, may drive land conversion for agricultural use. While the existing evidence from similar contexts in Southeast Asia suggests that this might not be a concern in Cambodia (Ferraro & Simorangkir, 2022), the need to assess indirect impacts on environmental degradation still remains (Alix-Garcia et al., 2013; Gilliland et al., 2019). Therefore, it may be worth complementing agricultural risk reduction with programs promoting environmentally sustainable methods of agricultural intensification (Tilman et al., 2017).

As the relationship between poverty alleviation and environmental degradation is likely to be context dependent, so is the broader applicability of transfer payments to reduce wildlife hunting. Wherever a negative relationship exists beyond rural Cambodia, wildlife conservation and poverty alleviation can be simultaneously pursued using cash transfers. Accordingly, these objectives can also be pursued in contexts of other forms of negative income shocks, including from other measures of agricultural production shocks and those caused by conflict. Our findings add to the limited, yet growing, literature of cash transfers that target limiting environmental degradation.

However, transfer payments are unlikely to be a panacea for wildlife conservation, particularly when hunted wildlife is a normal good. One such setting is likely to be in urban areas where poverty is lower, and demand is more responsive to cultural factors. While we find minimal evidence of urban households participating in hunting in Cambodia, demand for hunted wildlife in urban centres has been discussed in other parts of the world to be a growing threat to biodiversity loss. Similarly, the role of income in driving commercial hunting needs

to be better understood. Here, increasing income may allow for greater access to forest interior where wildlife availability is higher and also facilitate the use of improved hunting technology. Unsurprisingly, we find no suggestive evidence of commercial hunting in Cambodia as it is documented to be largely conducted by professional local hunting groups (Coad et al., 2019), international trafficking syndicates (Gray et al., 2018) and Vietnamese nationals in areas near the border (O'Kelly et al., 2018).

These final comments suggest that cash transfers are better understood as policies that need to be implemented in conjunction with (and not replacement of) a continuing focus on the protection of existing conservation areas and monitoring of bans on wildlife hunting and trade. Removing existing policies would only increase demand for wildlife hunting; we propose cash transfers to reduce the burden that these policies disproportionately place on the poorest populations. Comprehensive ex-post impact evaluation of cash transfer programs is a necessity.

By better considering trade-offs against human outcomes that, so far, have been largely neglected in the design of policy responses to the biodiversity crisis, we work towards achieving three Sustainable Development Goals simultaneously: No Poverty (Goal 1), Zero Hunger (Goal 2) and Life on Land (Goal 15). While the threat of wildlife hunting remains a pressing issue, with potentially irreversible consequences of biodiversity loss and severe impacts on the livelihoods of vulnerable populations, urgent action is required to better understand the drivers of wildlife hunting and ensure that approaches to conservation are both equitable and effective.

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