Demography of *Quasipaa* frogs in China reveals high vulnerability to widespread harvest pressure

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ABSTRACT

Over 20 species of amphibians are being harvested in Asia with local consumption exceeding the volume of international export in some countries. Over-harvesting is a particular concern for large-bodied species as their life history traits often make them prone to population depletion. *Quasipaa spinosa* is a large, stream-dwelling frog that is considered a delicacy in China. The demand for its meat is high, with domestic trade amounting to USD 32 m in one province alone in one year. Although it is known that populations of *Q. spinosa* have been heavily depleted, no information on the demography of wild populations has been available to assess declines and vulnerability to ongoing exploitation. We studied *Q. spinosa* in Hong Kong, China, where populations are protected from hunting and habitat is largely unaltered. Results from a population viability analysis suggest that these populations are highly vulnerable to hunting activity; hunted populations were associated with much lower expected abundance and substantially higher risk of extirpation than populations free from harvesting pressure. Harvesting confers at least 59% reduction of abundance in 5 years. Based on ongoing harvesting of this species from the wild and our determination that relatively low levels of harvesting can lead to population extirpation in the short term, we call for the establishment of protection measures (State Protected Animal status) for this species throughout China and advocate for strengthened enforcement of existing regulations in nature reserves and other protected areas. Some currently harvested amphibians in China, including other species of *Quasipaa*, are probably similarly threatened by unregulated harvesting activities and we recommend that their status and vulnerability to harvesting be carefully reviewed.

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1. Introduction

Harvesting of wildlife has emerged as one of the most important global threats to vertebrate populations (Bennett et al., 2002; Hoffmann et al., 2010). Although a broad array of wildlife species are hunted, amphibians remain an important food resource in many countries. It is estimated that over one billion frogs are traded internationally every year to meet the demand for frogs' legs in the US and Europe, and especially in France and Belgium (Gratwicke et al., 2009; Mohneke et al., 2009; Altherr et al., 2011; Warkentin et al., 2009). Most frogs are imported from Southeast Asia, as France and India have banned collection of native frogs (Altherr et al., 2011). Nearly half of frogs traded internationally come from Indonesia (Gratwicke et al., 2009). Between 2000 and 2009, the European Union imported a total of 46,400 tons of frogs' legs mainly from Asia, which amounts to an average of 93–230 million frogs annually (Altherr et al., 2011). High demand for frog meat has severely depleted populations of multiple species worldwide (Jensen and Camp, 2003; Carpenter et al., 2007). Over 20 species are harvested on a commercial scale in Asia (Kusrini and Alford, 2006; Altherr et al., 2011), and populations of these species are declining as a result. For instance, of the 39 ranid frogs harvested in China, populations of 12 of them are considered to be rapidly declining (Carpenter et al., 2007). Two species from Asia are now listed on CITES Appendix II and 16 more were proposed for listing in 1992 (Altherr et al., 2011). Amphibians generally have short lifespans, high fecundity, and large population sizes. These life history features, in part, have prompted researchers to conclude that amphibians are resilient to harvesting (Lau et al., 2008), even though annual take of amphibians currently numbers in the hundreds of millions (Kusrini, 2005).

Although international trade of frog meat is well documented, the extent of amphibian take for local consumption remains understudied, and in some cases, the quantities harvested for local consumption may far exceed the quantities exported. For instance, it is

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estimated that the number of frogs harvested and consumed within Indonesia may be two to seven times the quantity exported (Kusrini, 2005). Similarly, revenues from global trade of frog meat has been estimated as USD 40 m annually (Gratwicke et al., 2009), yet domestic trade in China of a single species, the giant spiny frog (*Quasipaa spinosa*), is estimated to be USD 32 m in one year from a single province (Yu and Ma, 2012). Local consumption alone can severely deplete populations as has been reported in West Africa (Mohneke et al., 2009) and Peru (Angulo, 2008).

Populations of *Q. spinosa* are estimated to have declined by more than 30% over the past decade, likely as a consequence of overharvesting (Lau et al., 2004). This species is now listed as Vulnerable by the IUCN and the China Red List (Zhao, 1998; Lau et al., 2004). Despite ongoing population declines, the species has not been designated as a State Protected Animal by the Chinese Government, and thus there are no regulations for its protection except where prohibitions on collection are enforced within China's nature reserves. This species is an important food and medicinal resource in China, (Ye et al., 1993; Gao, 1996), and thus we sought to assess the impacts of harvesting on populations. In this paper, we quantify the demography and related aspects of the ecology of *Q. spinosa* populations in Hong Kong, and we present the first quantitative assessment of the population-level consequences of hunting and commercial harvest on this widely harvested amphibian species. We suggest that there is an urgent need for similar studies on other Asian amphibian species potentially threatened by overharvesting.

2. Materials and methods

2.1. Study sites

We studied populations of *Q. spinosa* in two streams in Tai Mo Shan Country Park, a protected area with minimal human disturbance. Stream A (750–850 m in elevation) and Stream B (600–700 m) were not connected. Tai Mo Shan, the tallest (960 m) mountain in Hong Kong, is characterized by steep slopes with high gradient streams. Surrounding vegetation is grassland and shrubland, with patches of montane forest in ravines.

2.2. Animal sampling

On each stream, we surveyed a 200-m transect. Transect locations were selected based on known occurrences of the species on each stream. Transect length was limited by sections of dangerously steep terrain that bounded locations of known occurrence. Surveys were conducted every one to two months from September 2009 through July 2012, which included wet (May to October) and dry (November to April) seasons. Two to three surveyors walked upstream, one hour after dark, searching for frogs. All *Q. spinosa* large enough to be marked (snout-vent length, SVL $\geqslant$ 40 mm)

were captured. We recorded SVL (mm), weight (g), sex (males have keratinized spines on chest and fingers), and location of frogs on transects. Frogs were classified as unsexed juveniles (40 mm $\leqslant$ SVL $\leqslant$ 89 mm) or adults (SVL $\geqslant$ 90 mm) (Liu et al., 1990). Each new individual was injected with a passive integrated transponder tag (Biomark, Boise, Idaho, US). Recaptured frogs were identified by scanning the frog with a tag reader. We recorded search time for each survey, and search effort was standardized among surveys.

2.3. Population vital rates

We used Cormack–Jolly–Seber (CJS) models to estimate annual survival (ϕ) and recapture probability (ρ), and closed population models (jackknife estimator; Burnham and Overton, 1978) to estimate annual abundances (N). Survey data were pooled into three-month intervals, resulting in 12 pooled survey bouts over the three-year study. Abundance was estimated separately for the two populations, and data from the two sites were pooled for survival estimation. Sex/age class (adult male, adult female and juvenile), survey bout, study population and season were included as covariates in the candidate CJS model set for modeling survival (ϕ) and capture probability (p). Of those models for which all parameters were uniquely estimable, the five models with lowest AIC$_c$ were retained for model averaging (Burnham and Anderson, 2002). All analyses were performed using Program Mark 7.1 (White and Burnham, 1999).

2.4. Population viability analysis

Population viability analysis (PVA) was conducted to assess the vulnerability of *Q. spinosa* to hunting activity. Biological parameters for PVA were based on Fei et al. (2009) and this study. Models were run for 100 years, and risk was summarized as the percentage of simulation replicates that were extirpated. Capture efficiency of adults for each survey (subsequently used for assessing harvest impacts) was estimated as the number of adults captured per survey bout expressed as a proportion of total estimated abundance at each stream. Harvest was modeled as the annual removal of mature individuals by a single collector assuming either (1) moderate capture efficiency (mean capture efficiency based on abundance estimates; hereafter, moderate hunting scenario) or (2) high capture efficiency (95% confidence limit of capture efficiency based on abundance estimates; hereafter, intense hunting scenario). To test the population-level impacts of harvest while accounting for substantial parameter uncertainty, six scenarios were created, comprising all possible combinations of three harvest scenarios (no hunting, moderate hunting, intense hunting) and two recruitment scenarios (moderate recruitment and high recruitment). Moderate recruitment was estimated as the expected juvenile abundance (via jackknife estimator, averaged across the two study sites) in year *t*, divided by the expected abundance of reproductive

Table 1
Cormack–Jolly–Seber model estimates of apparent survival (ϕ) for *Quasipaa spinosa* in two streams in Hong Kong, China.

Site	Season	ϕ	SE	95% CI	Estimated by
Stream A	Annual	0.434	0.053	0.335–0.540	ϕ(site)$p(t)$
	Dry	0.441	0.098	0.265–0.632	ϕ(season * site)$p(t)$
	Wet	0.428	0.125	0.216–0.670	ϕ(season * site)$p(t)$
Stream B	Annual	0.696	0.073	0.538–0.818	ϕ(site)$p(t)$
	Dry	0.940	0.154	0.075–0.999	ϕ(season * site)$p(t)$
	Wet	0.476	0.133	0.242–0.720	ϕ(season * site)$p(t)$
Overall	Annual	0.518	0.048	0.424–0.609	ϕ(site)$p(t)$
	Dry	0.552	0.799	0.336–0.750	Models averaged
	Wet	0.492	0.064	0.345–0.640	Models averaged