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Spatial variation in the willingness to accept payments for conservation of a migratory wildlife corridor in the Athi-Kaputiei Plains, Kenya

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ABSTRACT

To be effective in promoting the conservation of migratory wildlife, recipients of payment for ecosystem services (PES) must be willing to accept payment along the entire migratory corridor. This paper investigates spatial variation in willingness to accept (WTA) payments made by the Wildlife Conservation Lease Program in the Athi-Kaputiei plains of Kenya. The program, designed as an incentive to keep land open for wildlife and livestock, offers land owners 10 US$ per ha per year, irrespective of location. We model the relation between WTA and distances to roads, towns and rivers, annual precipitation and slope and display the predicted spatial variation in WTA. The results reveal significant spatial variation in willingness to accept payments for availing land for conservation, with higher WTA concentrated away from roads and also in the Southeast of the plains. The results further suggest that wildlife movement will be blocked due to low WTA in the proximity of towns and tarmacked roads. We conclude that an effective strategy to keep the land open for migratory wildlife should consider spatial variation in WTA payment for land lease. It is suggested to consider stratifying the lease rates geographically to reflect the underlying spatial variation in WTA.

1. Introduction

Ecosystems provide highly valuable services (Costanza et al., 1998), but are rapidly deteriorating, with adverse consequences for human well being, especially for the poor in developing countries (Millennium Ecosystem Assessment, 2003). Payment for ecosystem services (PES) has become a popular tool for managing ecosystems to safeguard and sustain their services, but is also increasingly being viewed as a potential pathway for reducing poverty in rural areas in developing countries (Bulte et al., 2008; Lipper et al., 2009; Pagiola et al., 2005).

The most widely used definition of PES currently is the one proposed by Wunder (2005): "PES is a voluntary conditional transaction with at least one seller, one buyer and a well-defined environmental service". Wunder (2005) pointed out that PES schemes are characterized by five salient criteria: a PES is (i) a voluntary transaction where, (ii) a well-defined ecosystem service is (iii) being 'bought' by a (minimum one) ecosystem service buyer, (iv) from an (minimum one) ecosystem service provider (v) if and only if the ecosystem service provider secures ecosystem service provision". There are a wide variety of schemes that use the term PES; not all of which satisfy all the five criteria, and thus are probably better termed PES-like schemes (Wunder et al., 2008).

The actual implementation of PES depends on the socio-political, economic and biophysical environments (Jack et al., 2008) and therefore is likely to vary with the prevailing socio-ecological context. Furthermore, the type of ecosystem service supported through PES can vary widely and range from carbon management, climate change mitigation, biodiversity conservation, landscape scenery and water

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management, to a “bundle” or mixture of these services (Wunder, 2005). Additionally, PES can also be distinguished by the ecosystem where it is implemented (forests, wetlands, croplands, rangelands); the source of funding (public versus private sector funding or a mixture of both); the land tenure (public, communally owned or private lands); and the scale of implementation (local, national and global) (Farley and Costanza, 2010).

PES in the form of direct payments is increasingly being adopted to promote the conservation of biodiversity especially in private lands outside protected areas (Ferraro and Kiss, 2002). An example of payments for biodiversity conservation within agricultural landscapes is the Regional Integrated Silvo-pastoral Ecosystem Management Projects (RISEMP) implemented in Colombia, Costa Rica and Nicaragua to support farmers to adopt silvi-cultural practices on their farms (Pagiola et al., 2004). PES for biodiversity can support conservation of individual species or management of ecosystems and landscapes (Milne and Niesten, 2009). In wildlife biodiversity conservation, PES is commonly applied to mitigate human-carnivore conflicts (Dickman et al., 2011; Nelson, 2009) or to prevent loss or deterioration of habitats critical to conservation by supporting conservation-friendly land uses (Milne and Niesten, 2009; Pagiola, 2003).

A critical aspect in biodiversity conservation is to ensure that land is managed to provide landscape connectivity (Rouget et al., 2006). Connectivity (Bennett, 1998) is an important characteristic of lands managed for biodiversity conservation in general, but is all the more important in the case of migratory mammals, such as wildebeest (*Connochaetes taurinus*) and zebra (*Equus burcheli*) populations in East African savannas, various antelope species in South Sudan, and the Mongolian gazelle (*Procapra gutturosa*) in East Asia, all of which migrate seasonally over distances as long as a hundred kilometers or more. These large mammal species are negatively affected by land uses that fragment landscapes, thereby impeding their migration and disrupting their life cycles that involve seasonal utilization of different parts of the landscape.

A number of PES schemes for wildlife have been developed to prevent the loss of critical habitats that serve as migratory corridors or dispersal areas for wildlife. Examples of PES schemes used to promote connectivity and the conservation of wildlife corridors and dispersal areas include the Meso-America Biological Corridor which spans eight countries from Mexico to Panama (Kaiser 2001) and the Terrat PES scheme in the Simanjiro plains, a key wildlife dispersal area for the Tarangire National Park in Tanzania (Nelson et al., 2010). Nevertheless, preliminary evaluation of PES for wildlife conservation in developing countries suggests limited effectiveness (Pattanayak et al., 2010), leading to concerns and calls for wider evaluation of their conservation impact (Ferraro and Pattanayak, 2006).

There are various approaches to assessing the effectiveness of PES schemes (Ferraro and Pattanayak, 2006). For example, Wunder et al. (2008) examined how effective PES programs have been at achieving their objectives of improving ecosystem service generation. They suggest four relevant criteria against which effectiveness of a PES program should be judged. First, all the potential service providers must enroll in the program. Second, providers must comply with the terms of their contract, a condition that introduces the need for verification and monitoring of compliance. Third, compliance must result in “additionality”; that is lead to a change in land use and service provisioning that would not have happened without the PES intervention. Fourth, the induced land-use changes must generate the desired ecosystem services.

From an economic perspective, the payments for delivery of an environmental service should create sufficient incentive to motivate the service providers to change their land use to deliver the desired ecosystem services. Delivering an ecosystem service typically involves investment, transaction costs and opportunity costs (Ferraro and Simpson, 2002), and the payment for the service should at least compensate for the ensemble of these costs. A key part of PES implementation is therefore how to determine the “price” of the ecosystem services desired (Barbier, 2011) to avoid over- or underpayments, which would lead to inefficiencies and ineffective outcomes (OECD, 2010; Wunder, 2007). Quantifying these various cost factors is difficult prior to or in the early stages of PES implementation. This is complicated by information asymmetries that allow providers to give an overestimated picture of their opportunity costs of providing ecosystem services (Ferraro, 2008). Increasingly, contingent valuation methods are being used to estimate the willingness to pay (WTP) on the side of ecosystem service buyers, and the willingness to accept payment (WTA) for ecosystem services providers (Rodriguez et al., 2011a, 2011b; Lewis et al., 2009).

The willingness to accept payment for an ecosystem service is the amount that a person is willing to receive as compensation to forego certain land uses on their land. The difficulty in estimating how much to pay, coupled with secondary objectives such as poverty alleviation in PES schemes means that many biodiversity PES schemes offer flat payments (OECD, 2010) that are homogeneous in space. This generates two problems leading to PES ineffectiveness. First, flat payments do not account for the heterogeneity of ecosystem services across the landscape (Wünscher et al., 2008). Second, it also does not account for spatial variation in opportunity costs among providers, which is both a function of landscape diversity, and the nature of PES schemes, which determines the level of the transaction costs; with ‘asset-building’ PES schemes requiring higher investments than ‘use-restricting’ PES schemes (Engel et al., 2008). These concerns concur with Lewis et al. (2009) who observed that voluntary incentive-based policies are often inefficient in achieving biodiversity conservation goals for entire landscapes, which arises primarily from the inability of regulators to control for variation of costs across landscapes.

This study aims to investigate the variation in willingness to accept payment for wildlife conservation in the Kitengela plains in Kenya. This is achieved by developing a model describing how this variation in WTA relates to distance to the nearest road infrastructure and other landscape variables, and implementing this model in a GIS environment to display the spatial variation of the willingness among land owners to accept the payment for the land lease implemented in the Kitengela plains. After introducing the Kitengela plains and stating the problems that motivated the development of the PES scheme in this area, we describe the methods used to collect data and model the willingness to accept payments for this land lease program.

2. Study area

2.1. Background

The 114 km^2 Nairobi National Park, one of the three urban National Parks globally, is located 7 km from the Centre of Nairobi metropolis (Rodriguez et al., 2011a, 2011b). The Park is fenced on three sides, but the southern boundary, marked by the Mbagathi river, is open and allows the movement of wildlife into private lands located in the 390 km^2 Kitengela and the larger 2456 km^2 Athi-Kaputiei plains (Nkendianye et al., 2009). When the Nairobi National Park was gazetted in 1946 it was recognized that it was too small to meet the ecological requirements of the migratory wildlife, which was still significant by then. The Kitengela plains and the Ngong Hills, which acted as drought refuges for wildlife, were thus declared as game conservation areas, but were never gazetted (Gichohi, 2003) Fig. 1.