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Analysis

Sustainability and the value of the 'regulating' services: Wetlands and water quality in Lake Victoria

Silvio Simonit*, Charles Perrings

Arizona State University, PO Box 874501, Tempe, AZ 85281, USA

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ABSTRACT

The regulating services provided by ecosystems are amongst the most important for the sustainability of resource use, and yet they are also amongst the least understood. This paper considers one set of regulating mechanisms – the buffering functions of wetlands – and considers the information needed to identify both the value of the services offered by wetlands, and their substitutes. Using data from a catchment discharging water and nutrients to the Kenyan segment of Lake Victoria, the Yala catchment, the paper models the interactions between agriculture and fisheries as mediated by wetlands at the lake margin. More particularly, it estimates the value of the forgone nutrient retention function involved in the conversion of the wetland to agriculture, and the scope for providing the same services through land use change elsewhere in the catchment. The total cost of the payments that would compensate farmers for on-farm nutrient buffering services is 3.86 M US\$year⁻¹, or 35% of the total gains from wetland conversion to crop production. This finding contributes to an understanding of both the value of regulating services and how they might most effectively be delivered in alternative ways.

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1. Introduction

The Millennium Ecosystem Assessment identified the regulating services as amongst the least understood but potentially most valuable services offered by ecosystems (MA, 2005). These services alter the reliability in the supply of provisioning services such as food and water. They do so by enabling ecosystems to continue to produce such things over a range of stresses or shocks, often of anthropogenic origin. Such services have been interpreted as providing 'insurance': not as cover against the financial consequences of loss, but maintenance of the capacity of the system to continue to function over a range of conditions (Loreau et al., 2002). This aspect of the sustainability of the flow of benefits from ecosystems is generally characterized as ecosystem resilience (Kinzig et al., 2006; Walker and Meyers, 2004). Since the transition from one ecosystem state to another generally involves crossing some threshold, the value of the regulating services provided by any ecosystem lies in its capacity to reduce the likelihood that will happen. More particularly, the value of the regulating services derives from the benefits they protect (see Goulder and Kennedy, 1997; Barbier, 2007; Barbier et al., 2009; Heal et al., 2005).

Amongst the most important examples of the regulating services described by the Millennium Assessment are those affecting water quality. Wetland ecosystems, including rivers, lakes, marshes, rice

fields, and coastal areas, are the source of a number of provisioning services that are sensitive to water quality. Inland fisheries are of particular importance in developing countries, and are sometimes the primary source of protein for rural communities. Nevertheless, capture fisheries are in decline worldwide. There are two main reasons for this. One is the fact that many fisheries are still effectively open-access resources, which results in the overexploitation of those resources. The other is the effect of land-based pollution. Many freshwater and marine capture fisheries have been severely affected by sewage, nutrients, synthetic organic compounds, sediments, metals, radionuclides, oil/hydrocarbons and polycyclic aromatic hydrocarbons (PAHs) (UN, 2004). The application of fertilizers in agriculture, fossil fuel burning, land clearance and biomass burning have long been recognised as a major source of the nutrient load in freshwater, coastal and estuarine systems (Oglesby, 1977; Nixon, 1988).

In this paper we focus on water purification and waste treatment services provided by wetlands through the filtration of nutrient runoff from agricultural land. Wetlands, and their relationship to both terrestrial and aquatic systems, have been thoroughly explored by hydrologists, ecologists, fishery scientists and limnologists amongst others (Lynne et al., 1981; Turner, 1982; Reckhow and Quian, 1994; Swallow, 1994; Mitsch and Wang, 2000; Trepel and Palmeri, 2002). Economists have tended to focus on their role as fish spawning grounds and nurseries for fry in coastal and estuarine systems (Swallow, 1994; Barbier and Strand, 1997). In this paper we focus on the capacity of wetlands to reduce the likelihood that land-based nutrient flows will lead to the eutrophication of oligotrophic lakes, or

* Corresponding author. Tel.: +1 480 727 7665; fax: +1 480 965 6899.

E-mail addresses: ssimonit@asu.edu (S. Simonit), charles.perrings@asu.edu (C. Perrings).

the further decline of eutrophic lakes. Given the value of the lake in some state, it is possible to derive the economic value of the buffering function of wetlands.

We model land–water interactions in a particular system – the Yala catchment discharging into the Kenyan part of Lake Victoria – with emphasis on the role of wetlands in the Yala estuary in protecting water quality and fishery productivity in the lake. Two processes have been found to be at work in existing studies of this phenomenon elsewhere. Nutrient enrichment has a positive effect on fishery productivity in nutrient-limited environments such as oligotrophic or mesotrophic lakes (Stockner and Shortreed, 1988; Melack, 1976a; Liang et al., 1981; Hoyer and Jones, 1983; Downing et al., 1990; Quirós, 1990; Gomes et al., 2002). However, there is also evidence that sustainable harvests of fish populations at upper trophic levels decline if the system becomes highly eutrophic (Beeton, 1969; Lee et al., 1991; Caddy, 1993). Excess nutrients affect fish productivity through changes both in the amount of food available (Bootsma and Hecky, 1993) and in the quality of the habitat (Hammer et al., 1993). Deoxygenated water boosts natural mortality of fish. Sedimentation negatively affects nursery grounds and may damage fish eggs. When combined with high fishing pressure, both effects can have a severe impact on fish stock biomass and fishery yields (Kemp et al., 2001).

In Lake Victoria, fish production in all three riparian states has grown dramatically since the introduction of the Nile perch (*Lates niloticus*) in the early 1960s. In the Kenyan waters of the lake output grew from around 17,000 tonnes per year in the 1960s to more than 200,000 tonnes in the early 1990s. During the 1980s, *Lates* catches increased exponentially rising in few years from virtually zero to almost 60% of total yield (Okemwa, 1984; CIFA, 1988; Ogari and Asila, 1990; Ogutu–Ohwayo, 1990; Ssentongo and Welcomme, 1985). However, from 1994 fish landings have been in sharp decline, mostly due to declining catches of Nile perch.

Several factors are implicated in this. Overfishing is one factor, but it is not the only one. Eutrophication, caused by nutrient runoff from agricultural land and discharges from urban settlements on the lakeshore is also an important factor. It stems from two things. One is an increase in nutrient loading due to the application of fertilizers on agricultural lands and the growth of human populations on the lakeshore. A second is a reduction in nutrient absorption by wetlands on the lake margins, largely caused by the conversion of wetlands to other uses. A number of water quality analyses over the last decades show that Lake Victoria has progressively shifted from a mesotrophic to a eutrophic state from both causes. Increasing chlorophyll-*a* concentrations have been reported (Ochumba and Kibaara, 1989; Gophen et al., 1995; Kenyanya, 1999; Lung'aya et al., 2000, 2001) against baseline values provided by Talling (1965, 1966) and Melack (1976b).

While the buffering function described here is not the only regulating mechanism involved in the maintenance of system resilience, it is representative of quite a large class of regulating mechanisms. Insights derived from the particular case of wetlands can, accordingly, help in understanding the value of regulating services in a larger set of systems. A critical feature of the problem is that the value of the regulating function of wetlands depends both on landscape characteristics at the source of pollution, and resource use at the sink. Previous economic research on externalities of this kind has focused on the relationship between upstream and downstream activities, with particular reference to nonpoint pollution (Xepapadeas, 1992; Shortle and Abler, 1997; Avila-Foucat et al., 2009). In this paper we adopt a spatially explicit approach to the analysis of off-site externalities of this sort. Each land user in a watershed produces a different level of externality according to the physical characteristics of the land. A spatially explicit approach makes it possible to estimate the spatial distribution of pollution externalities, and the value of wetland functions in terms of the

fishery it protects. This in turn makes it possible to tailor solutions to the characteristics of the watershed, so enhancing their efficiency.

2. Modelling Environmentally-mediated Interactions Between Sectors

We model the relation between land use change, wetland area, water quality and fish stock biomass. Our primary aim is to develop an approach to the valuation of this class of ecosystem service, and not to provide precise estimates for a specific system. Indeed, lack of long term monitoring data makes it impossible to calibrate a land use/water quality function on data for the particular system involved (Verschuren et al., 2002). Instead, we calibrate the model using data from a range of closely allied systems. This serves both to illustrate the data requirements of the approach, and to demonstrate its potential application to the evaluation of policy options.

The benefits of wetlands in this case are downstream economic activities (a fishery), the value of which depends on the effect of water quality on fish biomass productivity. Since all water and nutrients flowing from the upper basin drain into the lakeshore wetlands, the nutrient load in the lake is affected both by the extent and intensity of upstream agriculture and by the buffering function of the wetlands.

Given that we are interested in a specific aspect of the interaction between agriculture and the fishery – the effect of wetlands in mediating the fate and transport of agricultural pollutants – we abstract from all other sources of inefficiency in the fishery. At present both the fishery and the agricultural sector show the effects of poorly defined property rights. Specifically, the fishery is exploited under near-open access conditions, while the agricultural sector is suffering from increasingly negative soil nutrient balances induced by overexploitation of similarly open-access forest resources. To model the specific effects of biota on interactions between the sectors we control for both sources of inefficiency by supposing that each prioritizes the sustainability of the natural resources on which they depend, fish stocks and soil nutrients respectively. In particular, we assume that fishery operates at maximum sustainable yield (MSY—the espoused target of fishery managers) and that agriculture assures soil nutrient balance (recommended by agricultural extension agents). Neither assumption reflects current reality in the area, but their adoption makes it possible to isolate the effect of changes in vegetation on intersectoral nutrient flows (Fig. 1). Total phosphorus influences both agricultural output and the biological productivity of lakes and streams. In fact, the amount of phosphorus available is commonly the limiting factor in freshwater primary productivity. We model the impact of nutrient applications in agro-ecosystems, and nutrient flows into the lake ecosystem.

In agroecosystems, nutrient applications affect both harvested crops and vegetation cover, each of which affects the soil nutrient balance. In the lake ecosystem, nutrient flows affect the relative abundance of a number of species (wetland macrophytes, primary producers, and fish stocks). We first model the nutrient flows generated if farmers allocate resources – i.e. land and fertilizers – so as to maximize net benefits to themselves subject to a soil nutrient balance constraint: the private problem. We then consider the problem in which resources in the agricultural sector are allocated to maximize net benefits to all members of society: the social problem.

The stylized externalities of land management involve both an on-site effect on soil nutrients, which requires the use of chemical fertilizers to maintain the soil nutrient balance, and an off-site effect on the fishery sector that depends on the extent of the wetlands. To isolate the partial effect of land use on the fisheries (Fig. 1), we identify a number of other sources of nutrients in the lake. Atmospheric deposition may contribute up to 35.7% of the entire loading of phosphorus into Lake Victoria (Scheren et al., 2000). There are few local point sources, and these are accounted for, in Eq. (18),

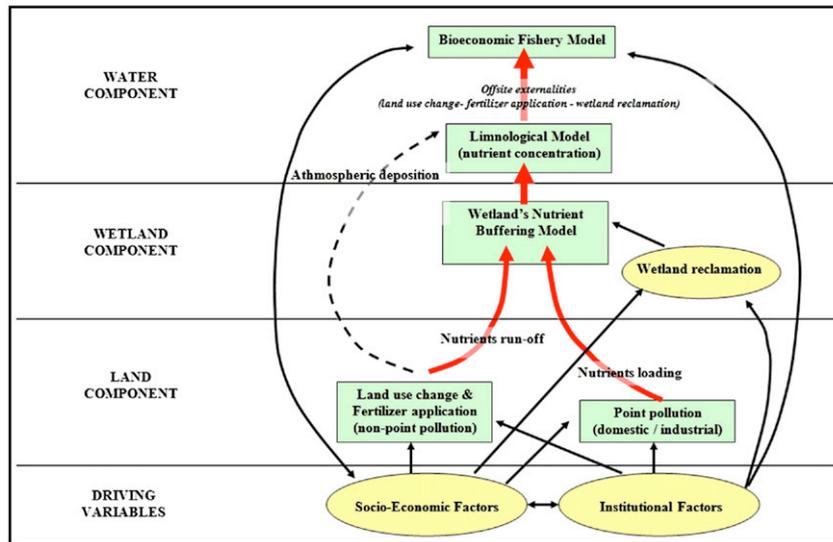


Fig. 1. Logical framework of the model.

within the average nutrient discharges from the rest of the Kenyan basin of Lake Victoria. A limnological model is required to transform nutrient loading into nutrient concentrations, which then feed into the bioeconomic fishery model.

The impact of nutrient loading on the water body is modelled through the effect it has on the productivity of the fishery. Considerable effort has been committed to modelling the dynamic effects of nutrients in shallow lakes (Carpenter and Cottingham, 1997; Carpenter et al., 1999; Janssen and Carpenter, 1999; Janssen, 2001). In this paper we focus on the direct effect on the growth of the fish population associated with the maximum nutrient outflows from the wetland. Changes in the state of the lake are accordingly observed as a decline in the productivity of the fishery.

To capture this effect we construct a bioeconomic model of the fishery that explicitly includes the impact of water quality. This approach that has been adopted in a number of studies (Ikeda and Yokoi, 1980; Fréon et al., 1993; Simonit and Perrings, 2005; Kasulo and Perrings, 2006). Following Simonit and Perrings (2005), we include a damage function that depends on nutrient loading from the watershed. In our case, changes in chlorophyll-a concentration – a measure of water quality and eutrophication – are the result of nutrient loading from the catchment, partially buffered by the wetland. In Lake Victoria there is evidence of total nitrogen being the limiting nutrient in offshore waters while total phosphorus concentration seems to be the key element influencing phytoplankton growth in inshore waters (Mugidde, 2001). Since our analysis is limited to the Kenyan side of Lake Victoria, which is mostly represented by inshore waters, phosphorous is the nutrient at issue.

The equation for fish stock dynamics in presence of nutrient flows and harvesting is:

$$\Delta X = X(X_t, D(z_t)) - h_{MSY}(z_t) \quad (1)$$

in which the growth function takes the specific form,

$$X(X_t, D(z_t)) = rX_t \left(D(z_t) - \frac{X_t}{K} \right) \quad (2)$$

where X_t represents the fish stock biomass; $D(z_t) \leq 1$ is the damage function in terms of nutrient loading z_t ; z_t is phosphorus loading ($t P year^{-1}$) from the watershed to the lake after passing through the wetland; $X(.)$ and $h_{MSY}(.)$ are the growth and harvesting functions respectively; and r is the rate of growth and K represents the carrying

capacity of the aggregate fish stock. More particularly, the damage function is represented by a negative quadratic function of chlorophyll-a concentration:

$$D(z_t) = W(z_t) - cW^2(z_t) \quad (3)$$

where W_t is chlorophyll-a concentration ($mg m^{-3}$) and c is an estimated coefficient. We accordingly assume that nutrient loading positively affects the growth of fish stocks up to a certain point, after which further increases in nutrient loading cause exponentially increasing losses. The environmental variable influences the growth rate at a given level of the stock (Simonit and Perrings, 2005).

To model the link between agriculture and the fishery sector, we assume that the effects of upstream activities are not reflected in market prices and are ignored by those responsible. The watershed area, which includes the sum of drainage land and wetlands, is denoted L_0 . This land is allocated between two uses: agriculture, L_A , and wetland, L_W . Land uses are assumed to be determined at the spatial unit of 1 ha. Within each hectare of land under agricultural use, land is either fallow, L_{Fi} (i.e. $\beta_i = 1$), in crop production, L_{Ci} (i.e. $\beta_i = 0$), or both (i.e. $0 < \beta_i < 1$), with $L_A = \sum_i (L_{Ci} + L_{Fi})$. In other words, farming households manage soil productivity by choosing agricultural intensity, β_i , proxied by the area left in fallow. It follows that:

$$L_A = \alpha_t L_0 \quad (4)$$

$$L_W = (1 - \alpha_t) L_0 \quad (5)$$

$$L_{Ci} = 1 - \beta_{it} \quad (6)$$

$$L_{Fi} = \beta_{it} \quad (7)$$

with,

$$0 \leq \alpha_t \leq 1; 0 \leq \beta_{it} \leq 1$$

where α_t is the percent of total land in the basin which is used for agriculture at time t , and β_{it} is the percent of the i th hectare of agricultural land that is in fallow at time t . So the share of land allocated to wetland in the watershed is given by α , and agricultural intensity by β . Since land and labor are assumed to be used in fixed proportions, choice of agricultural intensity also determines the allocation of household labor.

Land use externalities are modelled by introducing a nutrient runoff function for phosphorus. This function describes nutrient flows from the catchment into the wetland. In tropical soils, phosphorus is tightly bound by soil particles and runoff is represented almost entirely by soil erosion (Roy and Misra, 2003). We therefore take soil erosion, and the related sediment delivery, as a proxy for phosphorus loading. To simplify matters we do not model the dynamics of soluble phosphorus within river water. The general form of the nutrient runoff function is as follows:

$$g_t = \sum_{i=1}^{\alpha L_0} g_{it}(Q_{it}, P_{it}, \beta_{it}, \Omega_i) \quad (8)$$

where nutrient loading, g_t , into the wetland is expressed in $t P year^{-1}$ and represents the sum of the loading from each hectare; Q_{it} is a measure of soil phosphorus content in the top 20 cm of soil and expressed in $kg P ha^{-1}$; P_{it} is phosphorus fertilizer application ($kg P ha^{-1} year^{-1}$); and Ω_i is rainfall ($mm year^{-1}$). The specific form of the function estimated is:

$$g_t = \sum_{i=1}^{\alpha L_0} S_i A_{it}(\beta_{it}, \Omega_i) \delta \theta Q_{it} \quad (9)$$

where α_t and β_{it} are the already defined indices of land allocation in the catchment, and production intensity for the i th hectare of agricultural land respectively; A_{it} is a measure of soil erosion ($t ha^{-1} year^{-1}$); δ is a soil phosphorus enrichment factor; θ is the soil bulk density factor; and $0 \leq S_i \leq 1$ is the sediment delivery ratio (accounting for sediment deposition within the basin).

Soil phosphorus content depends on nutrient flows, crop production decisions and soil dynamics. The general form of the soil phosphorus function is:

$$\Delta Q_{it} = Q_i(Q_{it}, P_{it}, \beta_{it}, \Omega_i) \quad (10)$$

and the specific form estimated here is

$$Q_{it+1} = Q_{it} + P_{it}(1-\beta_{it}) + \phi_i(\Omega_i) + \chi \beta_{it} - \eta y_{it}(Q_{it}, P_{it}, \beta_{it}) - S_i A_{it}(\beta_{it}) \delta \theta Q_{it} \quad (11)$$

where,

$$y_{it} = \psi(1-\beta_{it})(a_1 + a_2 Q_{it} + a_3 P_{it} - a_4(Q_{it} + P_{it})^2) \quad (12)$$

is an agricultural yield response function, with coefficients a_1, a_2, a_3, a_4 , and unit conversion factor ψ . The variation in Q_i is expressed in $kg P ha^{-1}$; y_{it} is crop productivity ($t ha^{-1} year^{-1}$) for the i th hectare; χ is a constant representing the nutrient build-up factor due to fallow rotation ($kg P ha^{-1} year^{-1}$); η expresses the phosphorous content in the harvested crops ($kg P t^{-1}$); ϕ_i is atmospheric deposition of phosphorus ($kg P ha^{-1} year^{-1}$) on the i th hectare; and the other parameters are defined as above. P_{it} is the solution to a problem of the general form:

$$Max_{\alpha, \beta} = \sum_{t=0}^{\infty} \rho^t \pi_i(y_{it}(P_{it}, \beta_{it}), \omega_t) \quad (13)$$

subject to Eq. (11) and

$$P_i \geq \bar{Q}_i - Q_{it} \quad (14)$$

where π denotes net revenue, ρ is a discount factor and ω is a price vector. Land and labor allocations are both embedded in the choice of the intensity variable β_{it} . Specifically, we suppose that both land area (customary right) and labor (limited by family size) are worked in fixed proportions. It follows that the choice of β_{it} determines allocation of both. The soil dynamics in Eq. (11) represent a simplified

nutrient balance approach (Smaling and Fresco, 1993; Smaling et al., 1996) where the initial natural stock of phosphorus concentration in the soil is increased by fertilizer application on tillage land, L_{Ci} , nutrient build up due to fallow rotation, and atmospheric deposition. This stock, however, is diminished by losses both through the harvested crop and soil erosion. Crop harvesting is accordingly an important component of soil dynamics. Conversely, crop productivity is influenced not only by fertilizer applications, but also by topsoil quality (amongst other factors).

The specification of Eq. (12) employs a yield response function for fertilizer application on maize in Western Kenya (Mugunieri et al., 1997). Given that it derives from field research it may overstate actual yields, but it provides a reasonable estimate of productivity potential in the area.

The regulating function of wetlands is modelled through nutrient retention. A number of studies have analysed nutrient retention by wetlands as a function of wetland area and nutrient loads (Byström, 1998; Dortch and Gerald, 1995). We adopt the general mass balance model for phosphorus presented by Kadlec and Knight (1996), with the choice of $\alpha, 0 \leq \alpha \leq 1$, fixing the extent of the wetland. The specific form of the function applied here is:

$$z(g_t, \alpha_t) = g_t w(\alpha_t) = g_t \exp(-k(1-\alpha_t)L_0 / v) \quad (15)$$

where z_t is the nutrient outflow from the wetland to the main water body ($t P year^{-1}$); g_t is nutrient inflow to the wetland ($t P year^{-1}$) as determined by the runoff functions (8) and (9); $w(\cdot)$ is the nutrient retention per unit of nutrient loading, a function of wetland area; v is the water inflow to the wetland (m^3); k is the constant areal removal rate ($m year^{-1}$) for total phosphorus. Nutrient retention is the minimum rate achieved at the given wetland area. In other words, $z(g_t, \alpha_t)$ is the maximum nutrient outflow to the water body associated with the given value of α .

To get a measure of the value of a change in wetland extent for nutrient buffering, we need to identify the effect of interactions between the agricultural and fishery sectors along an optimal path. Subject to our assumption of the fishery operating under MSY management regime, the social problem requires maximization of net benefits across sectors through choice of the level of fertilizer application P , wetland extent α , and agricultural intensity β . The general form of this is:

$$Max_{\alpha, \beta} = \sum_{t=0}^{\infty} \sum_{i=1}^{\alpha L_0} \rho^t (\pi_{Ai}(P_{it}, \alpha_t, \beta_{it}, \omega_t) + \pi_{MSY}(P_{it}, \alpha_t, \beta_{it}, \omega_t)) \quad (16)$$

where ρ is the discount factor, π_{MSY} represents the benefit for the fishery sector operating at the MSY and π_{Ai} is the net benefit to the i th agricultural household. Benefits are approximated by net revenues in the two sectors. The primary control variable is the area of wetland, α . Optimizing this function subject to Eqs. (1) and (10) requires, amongst other first order necessary conditions, that:

$$\sum_{i=1}^{\alpha L_0} \frac{\partial \pi_{Ai}}{\partial \alpha_t} = -\rho \mu_{t+1} \left(\frac{\partial X}{\partial D} \frac{\partial D}{\partial z} \frac{\partial z}{\partial w} \frac{\partial w}{\partial \alpha_t} \right). \quad (17)$$

The term on the right hand side of Eq. (17) is the value of the nutrient buffering function of the wetland and $\rho \mu_{t+1}$ represents the discounted user cost of the exploited fish stock. Along an optimal trajectory, condition (17) implies that the marginal benefit of wetland depletion, measured in terms of the additional net benefits of agricultural production, should be equal to the marginal cost of wetland depletion measured in terms of the value of the forgone nutrient buffering services. Specifically, this is the value of the marginal impact of a change in wetland area on the growth of fish stocks, given nutrient inflows to the wetland. The marginal impact depends both on the effect of wetland extent on nutrient retention, $(\partial z / \partial w)(\partial w / \partial \alpha_t) = g_t(\partial w / \partial \alpha_t)$, and on the damage associated with

changes in nutrient loading, $\partial D/\partial z$. The value of this latter term depends on the state of the system. If the system is initially in an oligotrophic state, then the marginal damage of wetland depletion will be positive. Indeed, at the point where nutrient loads push the system from an oligotrophic into a eutrophic state, the marginal damage of a change in the extent of the wetland may be very high. If the system is initially in a eutrophic state (the damage has already been done), further changes in wetland extent may have little or no marginal impact on fish stocks. The value of this marginal impact at time t – the discounted user cost, $\rho\mu_{t+1}$, of the exploited fish stock – depends on the solution to the optimal fish harvesting problem, which in turn depends on the market price of fish, institutional conditions, access rights, the regulatory regime and the fishers' objectives.

3. Methods: Estimating the Value of Nutrient Buffering in the Yala Wetlands, Kenya

We estimated the value of the forgone nutrient buffering function of wetlands proposed for conversion in the Yala Swamp and its catchment, on the Kenyan side of Lake Victoria (Fig. 2). The Yala river discharges into Lake Victoria through the wetland. Yala Swamp has already been significantly reduced by 'reclamation' for agricultural development since the mid 1960s, and an additional area of 9200 ha from the remaining 15,200 ha of wetlands was demarcated for conversion to agricultural use in the Lake Basin Development Authority Five Years Plan 1989–1993 (Mwakubo et al., 2004). We took this proposal as the basis for the change in the extent of the wetland to be valued.

3.1. Damage Function Estimation

To obtain an estimate of the damage function, Eq. (3), we linked nutrient loading to phytoplankton density (i.e. chlorophyll-a concentration), drawing on the phosphorus mass balance models first developed by Vollenweider (1968, 1969). Without the support of specific studies for Lake Victoria, we exploited the literature on chlorophyll/nutrient relationships in freshwater ecosystems more generally (Sakamoto, 1966; Megard, 1972; Dillon and Rigler, 1974; Jones and Bachmann, 1976; Schindler, 1978; Straškraba, 1980; Smith, 1982; Prairie et al., 1989). We selected the general equation proposed by Dillon and Rigler (1974) to describe the dependence of the overall phytoplankton biomass on phosphorus concentration in the water (see Table 1). Phosphorus loading from the Yala basin was augmented

by atmospheric deposition and nutrient loading from the rest of the Kenya watershed of Lake Victoria and then converted to a concentration. Using this, we were able to estimate the marginal impact on the growth of fish stocks from the nutrient loading of each i th ha of land in the Yala watershed. Specifically, we estimate the marginal impact on the environmental variable W , chlorophyll-a concentration, from nutrient loading, g_{it} , from wetland area and from a set of constants describing long term average conditions in the lake:

$$W_{it} = \omega \left(\frac{g_{it}(Q_{it}, P_{it}, \beta_{it}, \Omega_t) \exp(-(k(1-\alpha_t)L_0)/v) + J + N}{U + \frac{sV}{H}} \right)^\varphi \quad (18)$$

where J is the long term average atmospheric deposition on the lake's surface ($t P year^{-1}$); N is the long term average phosphorus loading from the Kenya basin of Lake Victoria other than Yala watershed ($t P year^{-1}$); V is the lake volume (m^3); U is the hydraulic superficial outflow from the lake ($m^3 year^{-1}$); H is the mean depth of the water body (m); s is the settling velocity rate for phosphorus ($m year^{-1}$); ω and φ are estimated parameters of the model, and other variables and parameters have already been defined.

3.2. Spatial Modelling of Nutrient Loading

We next delineated the Yala watershed and estimated soil erosion. Using the ArcGis 9.2 application and a Digital Elevation Model (DEM) of Kenya based on the 1:250,000 contour map provided by the International Livestock Research Institute (ILRI), we defined the extent of the Yala watershed and its river network. We then estimated soil erosion and the associated sediment delivery that are key factors in determining nutrient loading. Using the Universal Soil Loss Equation approach (Wischmeier and Smith, 1978) and assuming no soil management to reduce soil erosion in the Yala basin, we obtained the soil erosion grid from map overlay of rain erosivity, soil erodibility, length-slope, and land cover factors respectively. In this approach, the rain erosivity layer is determined from a mean annual precipitation map (ILRI), adopting a functional relationship (Lufafa et al., 2003) estimated for a microcatchment study in Uganda. The latter uses coefficients derived by regressing long term rainfall against erosivity values determined by Moore (1979) for the Lake Victoria region. The length-slope grid is computed from the DEM using the approach developed by Moore and Burch (1985) which considers the upslope contributing area per unit width of contour (or rill). The soil erodibility layer was obtained from a soil characteristics map of

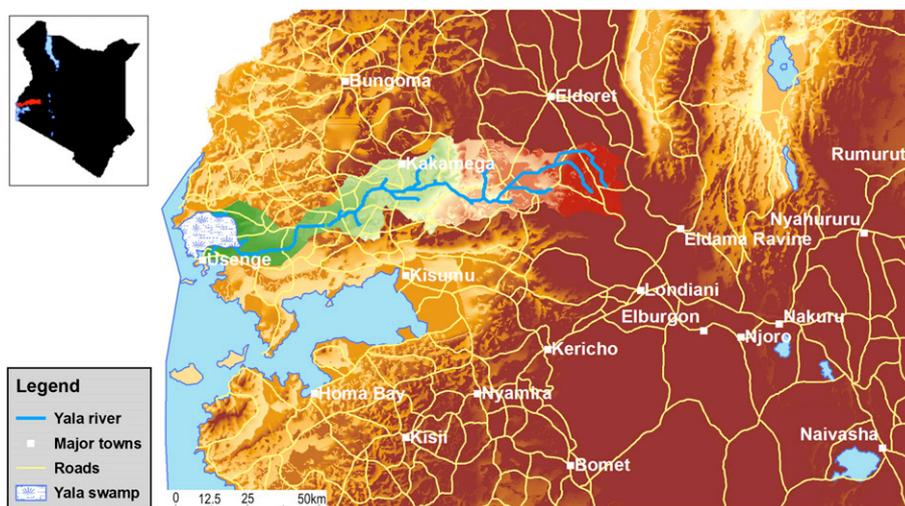


Fig. 2. The Yala watershed.

Table 1
Values for the parameters used in the model.

Variable	Description	Value	Units	Source
W_t	Chlorophyll-a concentration in the lake's waters	$W_t = 0.0731\bar{P}_t^{1.449}$	mg m^{-3}	Dillon and Rigler (1974)
\bar{P}_t	Total phosphorus concentration in the lake's water	$\bar{P}(z_t) = ((z_t + J)10^0) / (U + (s/H)V)$	mg P m^{-3}	Vollenweider (1968, 1969)
s	Settling velocity for total phosphorus	16	m year^{-1}	Chapra (1975)
v	Water inflow to the Yala wetland	1114×10^6	$\text{m}^3 \text{ year}^{-1}$	Balirwa and Bugenyi (1988)
k	Areal removal rate constant for total phosphorus	12	m year^{-1}	Kadlec and Knight (1996)
U	Hydraulic outflow from the lake (Kenyan side)	0	$\text{m}^3 \text{ year}^{-1}$	Our assumption
V	Average volume of the lake (Kenyan side)	5.3664×10^{10}	m^3	Our assumption
H	Mean depth of the lake (Kenyan side)	12	m	Our assumption
J	Atmospheric deposition of phosphorus on the lake's waters (long term average, Kenyan side)	1460	t P year^{-1}	COWI (2002)
N	Superficial loading from the Kenya watershed of Lake Victoria except Yala basin	1823	t P year^{-1}	COWI (2002)
L_0	Watershed extent	309027	ha	GIS estimation
$Q_{it=0}$	Phosphorous soil concentration (initial value at $t=0$)	30.25	kg P ha^{-1}	Onyango (1994)
A_{it}	Soil erosion	$A_{it} = R_i G_i L S_i C_{it}$	$\text{kg ha}^{-1} \text{ year}^{-1}$	GIS estimation based on Wischmeier and Smith's (1978) USLE
R_i	Rain erosivity factor	$R_i = 47.5 + 0.38\Omega_i$	-	GIS estimation based on Lufafa et al. (2003)
Ω_i	Mean annual rainfall	Rainfall distribution map	mm year^{-1}	International Livestock Research Institute (ILRI)
G_i	Soil erodibility factor	Soil erodibility map	$0 \leq G_i \leq 1$	GIS estimation based on ILRI's soil classification map and Wischmeier and Smith's (1978) parameters
LS_i	Length/slope factor	$LS_i = (\text{upslope contributing area}/22.13)^{0.4*} (\sin \text{ of slope angle in degrees}/0.0896)^{1.3}$	$0 \leq LS_i \leq 1$	GIS estimation based on ILRI's digital elevation model and Moore and Burch (1985)
C_{it}	Land cover factor	$C_{it} = 0.01\beta_{it} + 0.7(1 - \beta_{it})$	$0 \leq C_i \leq 1$	GIS estimation based on ILRI's tree cover percent map and Wenner's (1981) parameters
S_i	Sediment delivery ratio	$S_i = 0.4724 (\text{catchment area})_i^{-0.125}$	$0 \leq S_i \leq 1$	GIS estimation based on ILRI's digital elevation model and Vanoni (1975)
r	Intrinsic growth rate of the fish stock	0.201033	-	Simonit and Perrings (2005)
K	Carrying capacity of the fish stock	53082	-	Simonit and Perrings (2005)
c	Impact coefficient for chlorophyll-a concentration	0.029214	-	Simonit and Perrings (2005)
p	Unit price of the aggregate fish stock	744.29	$\text{US}\$ \text{ t}^{-1}$	Kenya Marine and Fishery Research Institute (KMFRI)
ϕ_i	Atmospheric deposition of phosphorus on land in Yala watershed	$\phi_i = 0.053\Omega_i^{0.5}$	$\text{kg P ha}^{-1} \text{ year}^{-1}$	GIS estimation based on ILRI's rainfall distribution map and Stoorvogel and Smaling (1990)
χ	Phosphorus build-up factor due to secondary forest fallow rotation	11	$\text{kg P ha}^{-1} \text{ year}^{-1}$	Nye and Greenland (1960)
η	Crop nutrient content	9.4	$\text{kg P/t of harvested product}$	Stoorvogel and Smaling (1990).
θ	Soil bulk factor	0.000379	-	Estimated from Ministry of Agriculture (1987)
δ	Soil phosphorus enrichment factor	1.5	-	Stocking (1984)
y_{it}	Maize crop productivity (agricultural production function)	$y_{it} = 10^{-3}(1 - \beta_{it})(924.6 + 60.806P_{it} + 31.866Q_{it} - 0.218(Q_{it} + P_{it})^2)$	$y (\text{t ha}^{-1}); P, Q (\text{kg P ha}^{-1})$	Adapted from Mugunieri et al. (1997)

Kenya (ILRI). The land cover grid was estimated by applying the land cover factor values – 0.01 and 0.7 for forest/savannah and maize cultivation respectively, reported by Wenner (1981) as representative for Kenya – to a digitalized map showing the percent of tree cover distribution in Kenya (ILRI). In other words, we considered tree cover rather than fallow distribution as a measure of agricultural intensity and assumed maize to be the representative crop in the Yala watershed, thus obtaining the spatial distribution of the parameter β in our general model. To account for the fraction of eroded soil that finally reaches the stream system or watershed outlet, we obtained a spatially distributed Sediment Delivery Ratio (SDR) grid using the functional relationship proposed by Vanoni (1975). We then multiplied that by the soil erosion grid to produce a sediment delivery map. For each pixel, this map estimates the amount of sediments (as proportion of the eroded soil) that would be expected to reach the basin outlet.

Most of the parameters for the model were obtained from the literature (described in Table 1). For the Kenyan sector of Lake Victoria we also assumed the hydraulic outflow to be fully compensated ($U=0$) by inflowing water from the Ugandan and Tanzanian side of the lake with the same concentration of phosphorus. This assumption is known in the engineering literature as the continuously stirred tank reactor (Reckhow and Chapra, 1983) and implies that the lake's waters are perfectly mixed.

To estimate the value of the nutrient buffering function of the wetland we assumed sufficient phosphorus fertilizer application to maintain the soil phosphorus balance (i.e. we assumed sufficient fertilizer applications to assure the sustainability of agriculture).

The fertilizer required to meet this balance condition varies for each hectare according to the natural conditions influencing soil erosion (site specific rain erosivity, soil erodibility and length/slope factors, and the sediment delivery ratio) as well the intensity of land use (β_i), which impacts both soil erosion through the land cover factor (C_i), and soil phosphorus dynamics. The 'sustainable' phosphorus application for each hectare was obtained by solving the soil nutrient balance Eq. (11) for P . The result is an expression for P_i as a function of β_i and the net soil erosion or sediment yield $S_i A_i$. We estimated the equation in a GIS environment through the raster calculator function by using percent tree cover and sediment yield maps as input grids, and assuming the initial soil phosphorus concentration of $Q_{it=0} = 30.25 \text{ kg P ha}^{-1}$ for each ith hectare. The resulting map identifies the spatial distribution of the amount of phosphorus fertilizer ($\text{kg P ha}^{-1} \text{ year}^{-1}$) required for agricultural sustainability (Fig. 3).

3.3. The Impact on the Fishery Sector

Given a constant nutrient inflow to the wetland, we then estimated the value of the nutrient buffering forgone by reducing

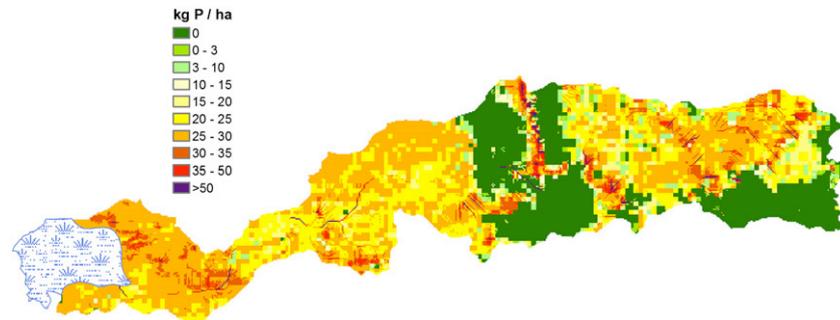


Fig. 3. Phosphorus fertilizer required ($\text{kg ha}^{-1} \text{ year}^{-1}$) for long term soil nutrient balance in the Yala basin.

the wetland extent by 60% (9200 ha) as in the proposed plan still under consideration by the local government. Taking the partial derivative of the fish stock with respect to α , as in the right hand side of Eq. (17), the estimated equation is:

$$\rho\mu_{t+1} = rX_t \left((.1164)10^{13} \left(\frac{W_t g_t}{M} \right)^{.449} - (.1871)10^{25} c \left(\frac{W_t g_t}{M} \right)^{1.898} \right) \quad (19)$$

$$\times \frac{W_t g_t k L_0 10^{-4}}{M v}$$

where,

$$M = U + \frac{sV}{H}$$

Using Eq. (19) we then derived the value of a change in wetland extent from the impact it has on the value of commercially exploited fish stocks. In the general case this depends on the harvesting strategy pursued by fishers, which varies with both access and management regimes (see Table 1 in Simonit and Perrings (2005) for the complete list of equilibrium solutions for the fishery under different management regimes). Given our initial assumption that the fishery is managed to achieve maximum sustainable yield (MSY), the value of the buffering function of wetlands was calculated assuming catches at MSY:

$$h_{MSY} = \sum_{i=1}^{\alpha L_0} h_{MSY_i}(P_{it}, \beta_{it}) = \frac{rK}{4} (W_{it}^2 (1 - cW_{it})^2). \quad (20)$$

We took the 2006 value of fish ($744.29 \text{ US}\$\text{t}^{-1}$), a change in wetland area of 9200 ha from an original 15,200 ha, and the parameter values given in Table 1, to calculate the impact of wetland reclamation in terms of loss to the MSY of the fishery. Since the MSY yield across the fishery also reflects a superficial loading (N) from the rest of the Kenya watershed of Lake Victoria, the change in nutrient load due to conversion of part of the Yala wetland is still small relative to the aggregate nutrient load. Note that this measures the impact of wetland reduction on the value of the variation of fish landings (Δh_{MSY_i}), assuming fertilizer applications (P_i) at the rate required to reach soil nutrient balance across the Yala catchment.

We found that if the wetland were reduced by 60%, the nutrient load “z” to the lake would increase from 34 t P year^{-1} to 92 t P year^{-1} . The loss of buffering functions would lead to an expected loss of 2666 t year^{-1} , equivalent to 1.98 M US\$. Dividing by the 9200 ha of wetland affected by ‘reclamation’ yields a cost in term of forgone fishery production of $216 \text{ US}\$\text{ha}^{-1} \text{ year}^{-1}$. This measure of the value of the regulating services of wetlands was then used to assess the efficiency of reclamation, and to evaluate the feasibility of alternative mechanisms for delivering the same regulating services. Assuming the maize maximum potential yield of $5.7 \text{ t ha}^{-1} \text{ year}^{-1}$ (Jaetzold and Schmidt, 1982) priced at $213 \text{ US}\$\text{t}^{-1}$, and assuming fertilizer applications of 95 kg P ha^{-1} at a cost of $256.5 \text{ US}\$\text{ha}^{-1}$, this implies a sustainable net benefit from conversion of $957 \text{ US}\$\text{ha}^{-1}$, or $741 \text{ US}\$\text{ha}^{-1}$ net of the fishery externality.

3.4. The Scope for a Solution Based on Payments for Ecosystem Services (PES)

We have already observed that the value of the buffering function of wetlands is sensitive to conditions both on and off shore. It is accordingly possible to identify the upstream conditions associated with changes in that value. In the case of Yala, for example, it is possible to identify the contribution of each hectare of land in the catchment to the nutrient load entering the wetland, and to use this information to identify policy options.

To illustrate, summing loads across the catchment, the baseline phosphorus loading “g” into the wetland is $176 \text{ t P year}^{-1}$. Using the GIS raster calculator with the percent tree cover (Fig. 5a) and phosphorus fertilizer (Fig. 3) maps as input grids, along with the price of fish ($744.29 \text{ US}\$\text{t}^{-1}$), we obtain the spatial distribution of the nutrient loading externality described in Fig. 4a. Note that the nutrient loading externality ranges from (virtually) zero for sites that experience relatively low soil erosion, to $\text{US}\$ 907 \text{ ha}^{-1} \text{ year}^{-1}$ for sites that lose a significant amount of sediment annually. Note also that the spatial distribution of the nutrient loading externality changes as a result of the wetland conversion (Fig. 4b). Areas that previously had no significant downstream externality now have an impact on the fishery, potentially up to $2465 \text{ US}\$\text{ha}^{-1} \text{ year}^{-1}$ for some sites.

There are two ways to use the spatially distributed externality data. First, if the externality is assigned by source (and not by the converted wetland area), we obtain an estimate of the additional damage done across the catchment as a result of the wetland reduction (Fig. 4b). In particular, by taking the difference between the two grids, we isolated the spatially distributed value of the forgone wetland buffering function as a result of ‘reclamation’, at an average of $32 \text{ US}\$\text{ha}^{-1} \text{ year}^{-1}$. This spatially distributed externality varies between farmers according to the use and physical characteristics of their land. Second, if the full value of the externality is assigned to the converted area (Fig. 4c), we obtained a different estimate of the damage associated with the conversion, $216 \text{ US}\$\text{ha}^{-1} \text{ year}^{-1}$. Note that these two approaches implicitly assume quite different property rights. If fishers were assigned rights to clean water, and if farmers on the converted wetland were assumed liable for the damage caused by nutrient runoff from elsewhere in the catchment, this is the amount they would have to pay in compensation for the reduction in nutrient regulation. If only farmers that were a source of nutrient runoff were assumed liable, the payments would vary but farmers on the converted land would carry no cost.

In fact, one reason why wetland conversion offers such a high potential return is that while it causes a reduction in nutrient buffering, it does not itself add to the nutrient burden since there is no silt runoff. Since nutrient retention is a function of vegetation cover either on farm or in the wetland, we can pose the question of whether on-farm nutrient buffering may be a viable alternative to wetland buffering. Of the total ($L_0 = 309,027 \text{ ha}$) land in the watershed, the

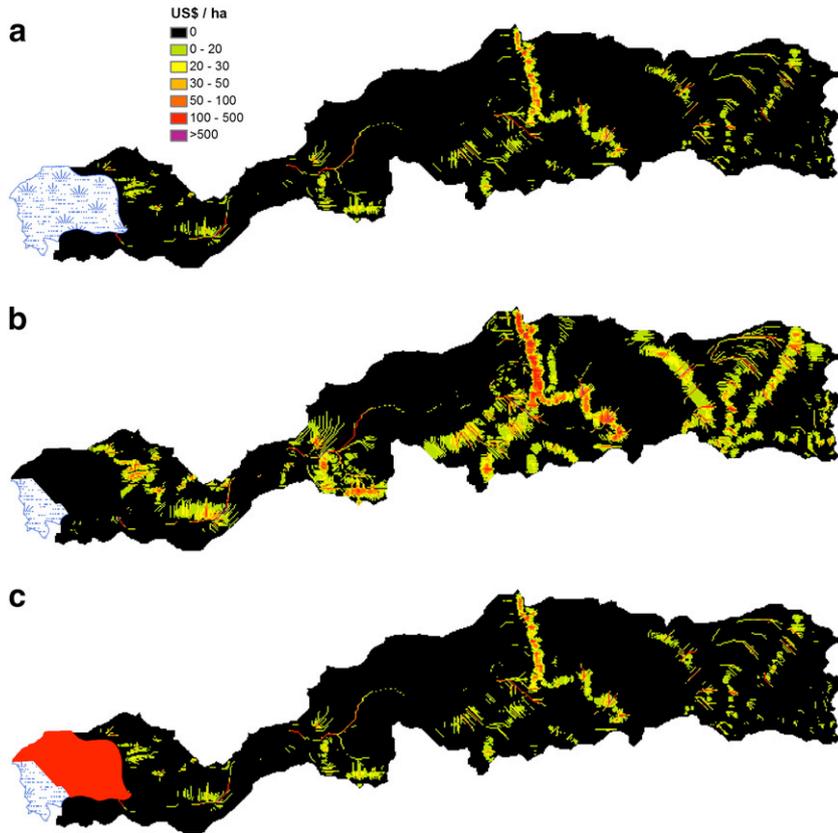


Fig. 4. Spatial distribution of nutrient loading externality before wetland reclamation (a) and after wetland reclamation, with externality assigned by source (b) or by the converted area (c).

area generating nutrient loading – and hence that could offer nutrient buffering through tree cover change – is 61,312 ha. Using an estimate of land cover (in terms of percent tree cover) required in order to neutralize the increased nutrient loading into the lake due to wetland reclamation, we found that it is feasible to decrease loading “g”

such that loading “z” remains at the same value as before wetland reclamation (Fig. 5b). Specifically, we found that the baseline phosphorus loading “g” of $176 \text{ t P year}^{-1}$ could be reduced to 65 t P year^{-1} by changing land cover. The value of $176 \text{ t P year}^{-1}$ is greater than the observed long term average loading of $102 \text{ t P year}^{-1}$

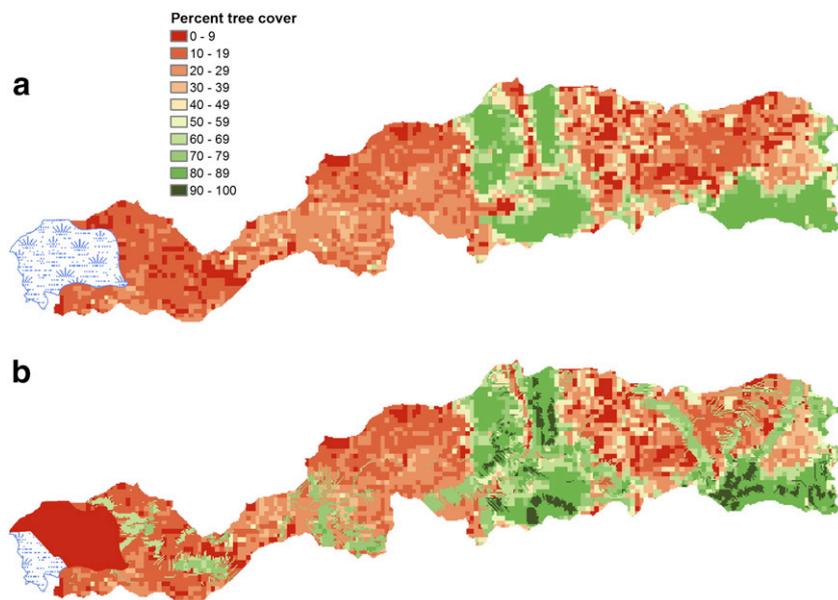


Fig. 5. Current distribution of percent tree cover (a) and percent tree cover required to compensating for reduced wetland buffering function (b).

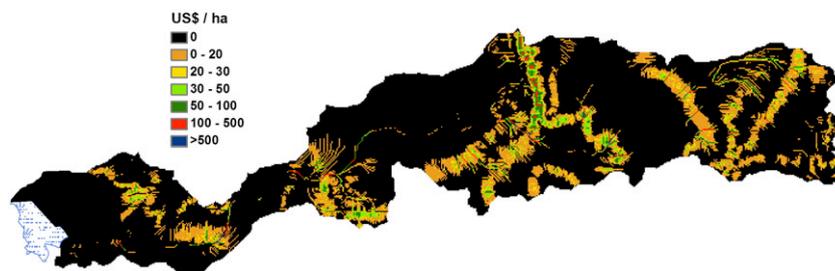


Fig. 6. Spatial distribution of payment for ecosystem services to farmers for reducing agricultural intensity by changing percent tree cover.

(COWI, 2002) due to the fact that we impose an agricultural sustainability constraint—requiring fertilizer applications to be consistent with reaching soil nutrient balance.

Fig. 6 then shows the payments to upstream farmers that would compensate them if they changed agricultural intensity on their plots to yield a constant nutrient load to the lake. In other words, this is the distribution of payments that would provide an on-farm substitute for the regulating services lost through wetland conversion. The total cost of the payments that would compensate farmers for on-farm nutrient buffering services is 3.86 M US\$ year^{-1} , or 35% of the total gains from wetland conversion to crop production. In principle it would therefore be feasible to implement a system of payments for ecosystem services that would be effective in offsetting the loss of nutrient buffering from wetland conversion.

4. Conclusions

Payments for ecosystem services schemes are an increasingly popular mechanism for internalizing the off-site of activities that confer benefits on others (Bulte et al., 2008; Engel et al., 2008; Ferraro and Kiss, 2002; Ferraro and Pattanayak, 2006; Wunder, 2007; Wunder et al., 2008). Implementation of such scheme requires information on the potential gains from alternative management strategies in order to identify the upper bounds of payments. One of the main benefits of modelling exercises of this kind is that they make it possible to test the efficiency of such policy instruments. The model of land-water interactions reported in this paper makes it straightforward to estimate the gains from alternative ways of delivering nutrient buffering on farm. The problem for policy in this case, as in many others, is that the value of regulating ecosystem services is rarely calculated and almost never factored into decision-making. It is important to understand the value of such services not just to assess the return to public investments that change those services, but also to identify and evaluate potentially corrective policies.

The value of the regulating services derives from the value of the provisioning services they protect. Since the sustainability of provisioning activities depends on the capacity of the system to function over a range of environmental conditions, the value of the regulating services will vary both with the value of the protected service and the variability of environmental conditions. In this paper we have taken the measure of this service to be the minimum nutrient retention provided, given prevailing climatic, soil, agricultural and institutional conditions. Other measures – based on acceptable risk or confidence intervals – might also be appropriate. In all cases, though, it is possible to identify the change in value associated with a change in conditions. For the Yala catchment we found that given the current state of the lake, and given our assumptions about behavior in both agriculture and the fishery, damage due to the loss of regulating services would not warrant conservation of the wetland. More particularly, we found that on-farm regulating services provided by vegetation could more effectively reduce the nutrient burden to the lake than wetland conservation. Of course, the fact that we have

abstracted from the institutional conditions under which both sectors currently operate means that the finding should not be taken to provide a reliable guide to policy in current conditions. However, it does illustrate the potential power of the modelling approach.

Finally, we emphasize that in order to undertake what is a controlled model experiment on the effect of biotic regulation of nutrient flows, we have ignored major existing sources of inefficiency in both agriculture and the fishery in the study area. The baseline against which we have evaluated the effects of changes in wetland extent is not the current regime in the fishery or agriculture, but a regime that is consistent with the sustainable use of both sets of resources. There is a strong (and well rehearsed) argument for addressing the main sources of inefficiency in both sectors. In fact without reform of the rules of access in the fishery, the potential gains offered by enhanced regulation of land-based pollution would likely be illusory.

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