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Evaluating indirect and direct effects of eco-restoration policy on soil conservation service in Yangtze River Basin

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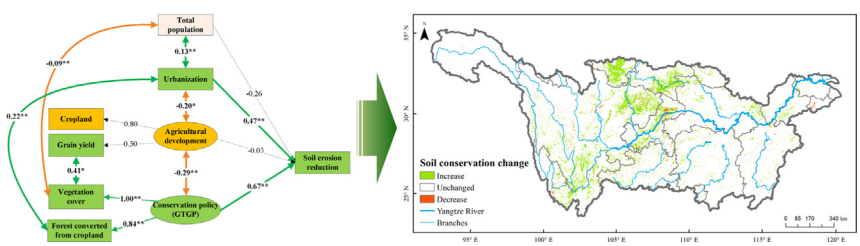
HIGHLIGHTS

- We assess the indirect and direct effects of Grain to Green Programme (GTGP).
- GTGP contributed significantly to an improvement in soil conservation services.
- GTGP enhanced soil conservation services significantly through faster urbanization.
- Vegetative cover and crop yields increased synergistically.
- Findings contribute to increasing such policies' effectiveness in the long run.

GRAPHICAL ABSTRACT



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ABSTRACT

The conservation impacts of policies that promote large-scale ecological restoration of ecosystem services and socio-economic development are well documented around the world. However, the effect of socio-economic development resulting from such policies on ecosystem services is rarely analysed, although it is important to do so if these policies are to be sustainable. We analysed the socio-economic impacts of soil conservation services from 2000 to 2015 in the Yangtze River Basin under the Grain to Green Programme (GTGP). Also we assessed the driving forces behind the programme: conservation policies, urbanization, agricultural development, and population growth. Our results show that during 2000–2015, cultivated area decreased by 7.5%, urban area increased by 67.5%, forest area increased by 2.1%, and soil erosion was reduced by 19.5%. The programme not only contributed significantly to an improvement in soil conservation services but also enhanced them significantly through faster urbanization. Furthermore, vegetation cover and crop yields increased synergistically, mainly due to high-efficiency agriculture that reduced the negative effect of the GTGP on agricultural production. Overall determining the indirect and direct effects of the GTGP on soil conservation and agricultural production are important for furthering our understanding of the long-term effects of ecological restoration policies, and the present study offers practical insights for ecological restoration of other watersheds.

1. Introduction

Soil erosion is a common global problem that leads to land degradation and the reduction of ecosystem services. In order to solve the problem, the strategies of rehabilitation and restoration of land-based ecosystems for recovering ecosystem services are adopted in many parts of the world (Keesstra et al., 2018). Soil management (Cerdà et al., 2017b; Parras-Alcántara et al., 2016), landscape solutions (Cerdà et al., 2017a), artificial strategies (e.g., check dams) (Vaezi et al., 2017), or the combination of artificial and natural solutions (Mekonnen et al., 2017) have been employed in some countries to mitigate regional soil erosion. Most of these strategies are considered

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nature-based solutions to land degradation problems. The global restoration movement is gaining momentum leading to the implementation of large-scale ecosystem restoration programs, such as large-scale forest and landscape restoration (Chazdon et al., 2017; Rodrigues et al., 2011), which have become increasingly popular policies for addressing land degradation (Aronson and Alexander, 2013). To counter ecological degradation often associated with rapid economic development, China has been implementing a number of national conservation policies often in the form of "payments for ecosystem services" (PES). The Grain to Green Program (GTGP) is one of China's largest programmes and is one of the most extensive PES programs in the world. The GTGP converts crops grown on steep slopes into forest land and grassland. The programme was launched in 1999. It was fully operational by 2000 and was fully extended to the vast majority of China (Liu et al., 2008). Assessing the socio-economic effectiveness of this substantial large-scale ecological restoration program is incredibly important for understanding how to generate comprehensive environmental policies.

The impacts of large-scale ecological restoration policies on ecosystem services and socio-economic development are well documented (Chazdon, 2008; Lü et al., 2012; Rodrigues et al., 2011; Xu et al., 2006). Some studies have shown that these policies can significantly increase the extent of ecosystem services (Benayas et al., 2009; Ouyang et al., 2016) and lead to a series of social-economic consequences (Cao et al., 2014; Elmqvist et al., 2015; Liu et al., 2008), and some studies have examined how the social-economic factors affect ecological conservation (Aradóttir et al., 2013; Timilsina et al., 2014; Yan et al., 2004). But the majority of studies only evaluate the direct relationships among ecological conservation policies, social-economic factors, and soil conservation service (Guerra et al., 2016; Haregeweyn et al., 2015; Ligonja and Shrestha, 2015). The effect of socio-economic development resulting from conservation policies focused on enhancing ecosystem services is rarely analysed, even though such analysis is important if these policies are to be sustainable.

The Yangtze River is considered the largest and the most important river in Asia (Yang et al., 2015). The GTGP began in the Yangtze River Basin where soil erosion in the upstream region is particularly severe (Long et al., 2006). These factors make the Yangtze River Basin an ideal study area where we sought to analyse the changes in soil conservation services from 2000 to 2015, and the driving forces (conservation policies, urbanization, agricultural economy, and population growth) behind these changes using structural equation modelling (SEM). The objectives of the study are: 1) determine changes in land cover and quantify soil conservation in the Yangtze River Basin from 2000 to 2015, and 2) evaluate the direct and indirect effects of the GTGP on soil conservation to determine practical policy implications.

2. Materials and methods

2.1. Study area

The Yangtze River Basin covers approximately 1,800,000 km^2, accounting for 18.8% of China's land area. The basin comprises of plains, plateaus, hills, and mountains, with the last three making up >80% of the total land area. Karst landform accounts for 18.8% of the total land area. Soil erosion is particularly severe in the upstream region (Long et al., 2006) where mountains occupy 50% of the landscape (Yan and Qian, 2004). The major land cover types are forest, cropland, and grassland (Fig. 1).

The Yangtze River Basin is one of the most densely populated and agriculturally productive areas in China. The Yangtze River Economic Belt, mainly located in the Yangtze River Basin, constitutes over 40% of both the national population and gross domestic product (GDP) (Chen et al., 2017), and has experienced a booming economy over the last decade (Zhang et al., 2014). Governmental agencies and departments have been working on ambitious development plans with clear targets for 2020 and 2030, aimed primarily at transport and urban development sectors in the Yangtze River Basin (Chen et al., 2017). In addition to urbanization, the Yangtze River Basin is a favourable location for agriculture, which accounts for 25% of the total cultivated land area in China. The agricultural production in the Yangtze River Basin accounts for 40% of the total value of the country's agricultural outputs.

2.2. Soil conservation service assessment

Soil conservation, in quantitative terms, is the difference between potential soil erosion and actual soil erosion (Rao et al., 2014). We used the Universal Soil Loss Equation (USLE) to calculate the soil conservation function, which is strongly influenced by rainfall, soil, topography, and vegetation. The model can be expressed as follows (Rao et al., 2014):

Potential soil erosion: $SE_p = \mathrm{R} \times \mathrm{K} \times \mathrm{LS}$.

Actual soil erosion: $SE_a = \mathrm{R} \times \mathrm{K} \times \mathrm{LS} \times \mathrm{C}$.

Soil conservation: $\mathrm{SC} = \mathrm{R} \times \mathrm{K} \times \mathrm{LS} \times (1 - \mathrm{C})$.

where SE_p is the potential erosion; SE_a is the actual erosion; SC is soil conservation capacity (expressed in tonnes per hectare per year); R is rainfall erosivity factor ($\mathrm{MJ \cdot mm \cdot ha^{-1} \cdot h^{-1} \cdot yr^{-1}}$) determined using kinetic energy (MJ) of raindrops and intensity of rainfall (in hectare-millimetres per hour) over one year; K is erodibility of the soil or the amount of soil lost through erosion per unit area following rainfall of a given intensity ($\mathrm{t \cdot ha \cdot h \cdot ha^{-1} \cdot MJ^{-1} \cdot mm^{-1}}$); LS is the topographic factor representing the effect of the length of slope; and C is the vegetation cover factor. In addition, an adjustment coefficient was used to correct for soil erosion in the widely distributed Karst areas (Rao et al., 2016).

The detailed calculation methods of the soil conservation service are detailed in Rao et al. (2014). First rainfall erosivity reflects the potential of raindrops and runoff to induce soil erosion (Wang and Jiao, 1996). We used mean annual rainfall erosivity at 603 weather stations, which we calculated using the Daily Rainfall Erosivity model (Yin et al., 2013). Next the kriging interpolation method was employed to obtain a raster layer for the R factor (spatial resolution of 90 m). Second soil erodibility reflects the sensitivity of soil particles to erosive forces, and is an internal factor affecting soil erosion that is closely related to soil attributes (Wang and Jiao, 1996). The Erosion/Productivity Impact Calculator (EPIC) was employed to calculate K using the soil clay, silt, sand and organic carbon content (Williams et al., 1983; Zhang et al., 2008). Third the topographic factor reflects the effects of terrain (slope length and gradient) on soil erosion (Remortel et al., 2001). We integrated the relevant research on gentle slopes and steep slopes, and performed calculations using different slope segments (McCool et al., 1993; Liu et al., 1994; Rao et al., 2014). The Digital Elevation Model (spatial resolution of 90 m) was the input data to calculate the LS factor. Fourth the vegetation cover factor describes the effect of vegetation on soil erosion, and is related to vegetation structure and cover. Values were assigned to the vegetation cover factor according to previous studies (Carter and Eslinger, 2004; Liu et al., 1999; Wei et al., 2002), where different ecosystem types and vegetation coverage were considered for forests, shrubs and grasslands. For farmlands (except paddy lands), the model established by Liu et al. (1999) was applied. For wetlands (including paddy lands, a type of farmland), cities, and bare lands (e.g., deserts, lichens), we used the following values 0, 0.01, and 0.7, which were derived from parameters using in Nonpoint Source Pollution and Erosion Comparison Tool (Carter and Eslinger, 2004). The input data of ecosystem classification images and vegetation cover in 2000 and 2015 were used to calculate the C factor.

Soil conservation is the reduced soil erosion by ecosystems, which are the estimates and not actual measurements of soil loss in our study. In order to validate the model's regional applicability, we validated the USLE model with soil erosion rates estimated from observed data for seven major river basins in China (the monitoring data of river sediment and the ratio of sediment transport from the Ministry of Water Resources of China) (Rao et al., 2014). The simulations (USLE) of soil erosion rates were significantly similar to corresponding