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Soil phosphorus budget in global grasslands and implications for management

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A B S T R A C T

Grasslands, accounting for one third of the world terrestrial land surface, are important in determining phosphorus (P) cycle at a global scale. Understanding the impacts of management on P inputs and outputs in grassland ecosystem is crucial for environmental management since a large amount of P is transported through rivers and groundwater and detained by the sea reservoir every year. To better understand P cycle in global grasslands, we mapped the distribution of different grassland types around the world and calculated the corresponding P inputs and outputs for each grassland type using data from literature. The distribution map of P input and output revealed a non-equilibrium condition in many grassland ecosystems, with: (i) a greater extent of input than output in most managed grasslands, but (ii) a more balanced amount between input and output in the majority of natural grasslands. Based on the mass balance between P input and output, we developed a framework to achieve sustainable P management in grasslands and discussed the measures targeting a more balanced P budget. Greater challenge is usually found in heavily-managed than natural grasslands to establish the optimum amount of P for grass and livestock production while minimizing the adverse impacts on surface waters. This study provided a comprehensive assessment of P budget in global grasslands and such information will be critical in determining the appropriate P management measures for various grassland types across the globe.

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1. Introduction

Grasslands are defined as “terrestrial ecosystems dominated by herbaceous and shrub vegetation and maintained by fire, grazing, drought and/or freezing temperatures. This definition includes vegetation covers with an abundance of non-woody plants and thus lumps together some savannas, woodlands, shrublands, and tundra, as well as more conventional grasslands” (White et al., 2000), making grasslands one of the most widespread biomes worldwide. Grasslands cover approximately 26% of the terrestrial area (Obermeier et al., 2016) and occur nearly on all continents except Antarctica (Rojas-Briales, 2015). Grasslands are most commonly found in arid and semi-arid zones (47% of the world's grasslands), followed by humid (23%), and cold (20%) regions (White et al., 2000). Grasslands encompass, for examples, the savannas of Africa and Australia, the cerrado and campo of South America, the prairies of North America, and the steppes of Central Asia (Kemp and Michalk, 2007). They cover about 117 million km^2 of vegetated lands, and provide forage for more than 1800 million livestock units and wildlife populations (Kemp and Michalk, 2007; Rojas-Briales, 2015). These vast areas of lands support over 800 million of human populations with abundant natural resources for the production of food, fuel, fiber and medicine (Kemp and Michalk, 2007; Rojas-Briales, 2015). By integrating the International Vegetation Classification (IVC) with the map of terrestrial eco-regions of the world, Dixon et al. (2014) classified the following nine grassland types: (1) Alpine scrub, forb meadow and grassland; (2) boreal grassland, meadow and shrubland; (3) tropical montane shrubland, grassland and savanna; (4) tropical freshwater marsh, wet meadow and shrubland; (5) tropical lowland shrubland, grassland and savanna; (6) Mediterranean scrub, grassland and forb meadow; (7) temperate grassland, meadow and shrubland; (8) cool semi-desert scrub and grassland; and (9) warm semi-desert scrub and grassland (Fig. 1). These grassland types vary with regards to different ecosystem properties, ranging from −0.3 °C to 27.4 °C, 299 mm to 1555 mm, 0.165 to 1.195, and 36 to 340 g m^{-2} for temperature, precipitation, aridity index (defined as precipitation/potential evapotranspiration), and aboveground productivity, respectively (Parton et al., 1995).

Soil factors (e.g., parent material, nutrient content, and organic matter), which determine phosphorus (P) content in grassland soils, are important contributors to the distribution of grassland types across the globe (Anderson and Talbot, 1965). The P content of soil parent materials, in particular, is closely related to total soil P because P is an element supplied by the parent material in natural ecosystems (Mage and Porder, 2013). With global phosphate rock depletion by biotic demands (e.g., agriculture), P gradually becomes one of the limiting elements of plant nutrients and vegetation productivity in terrestrial ecosystems, including in grasslands (Koppelaar and Weikard, 2013; O'Halloran et al., 2010; Schlesinger et al., 1990). In northeastern Brazil, cultivated semi-arid areas in the region experienced a decline in soil P due to mineralization of organic P and subsequent transformation of inorganic P surplus to unavailable forms rather than a net export of P to the crops (Tiessen et al., 1992). Accordingly, soil P then becomes an important limiting element which governs the accumulation of soil organic matter, which could differ within a particular climate zone (Tate and Salcedo, 1988). In colder artic fell field soils, for example, a smaller P mineralization occurred in contrast to in warmer sub-Alpine heath soils where P mineralization occurred in a larger quantity (Jonasson et al., 1993).

According to its availability for plant acquisition, P in the soil can be classified into three different pools: soluble P, labile P, and non-labile P (Larsen, 1967; Sharpley, 1995). The labile P pool, being a combination of mineral and organic P, is the main source of available P for plants (Sharpley, 1995), as the pool dissolves readily into soil solution by desorption from soil particles or by mineralization of organic P (Shen et al., 2011). The other two pools, on the other hand, are rarely used by plants. Only a small amount of soluble P is taken up by plant directly in orthophosphate or organic forms (McDowell and Sharpley, 2001). The non-labile P, despite their abundance (i.e., 95–99% of total P), is so inert and resistant to mineralization (Sharpley et al., 1995), and consequently it is released very slowly into the soil solution (Marschner and Rengel, 2007). All of these three pools, however, remain in a dynamic conversion by physicochemical reactions such as sorption and desorption (labile and soluble P), precipitation and dissolution (non-labile and labile P), mineralization and immobilization (inorganic and organic P) (Daly et al., 2015; Haygarth et al., 2005; Shen et al., 2011). They also share variable proportions in different soil types as various edaphic variables, such as soil temperature, moisture, pH, and plant depletion (Sharpley et al., 1995) determine their states. These dynamic conversions along with variability in soil properties contribute to the difficulties in accurately determining fertilizer P application rate, despite the presence of soil test P (STP), since the concentration of soil P varies with soil, plant species, and climatic factors, as well as analytical methodologies.

Considering the complexity of measuring P rate and its transfer (Haygarth et al., 2005), a reasonably simple approach (e.g., a nutrient mass balance) to facilitate the comparison of P loads across different grassland types is necessary as currently such information is critically lacking. A nutrient mass balance can be defined as an accounting approach for nutrient inputs and the outputs (Gourley et al., 2007). This is a useful technique to quantify and to predict nutrient deficit and surplus, although there are considerable assumptions associated with scaling up the measurements from different scales (e.g., plot, field, farm-gate, and farm-system scale measurements) by linear aggregation modeling, spatial statistics census, and remote-sensing approaches (Borbor-Cordova et al., 2006; Cobo et al., 2010; Gourley et al., 2007). In this review, P balance is calculated as the difference between inputs and outputs within pre-defined spatial domain, in this case different grassland types and the balance is expressed in the amount of P per unit area (Cobo et al., 2010).

Under the assumption that a positive balance indicates P surplus, representing potential loss to the ecosystem while negative balance indicates soil nutrient depletion (Cobo et al., 2010; Bouwman et al., 2013), we consider that ideally: (i) P inputs should be equal with the outputs to prevent soil P accumulation and depletion (Hooda et al., 2001; Ma et al., 2009) and (ii) P loss to environment should be lower than the threshold for environmental

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