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Long-term conservation agriculture influences ecosystem service in maize-wheat cropping system in the north-western Indo- Gangetic Plain

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ABSTRACT

As conventional agriculture faces several issues like water and labor crises, extensive land degradation, poor soil fertility, climate change, and increasing pressure on available agricultural land to population, in the recent, conservation agriculture (CA) is being promoted as a sustainable production technology. Many studies quantified ecosystem services (ES) in different agriculture practices but limited information is available on the impact of CA based practices on ES under maize-wheat cropping system of tropical agro-ecosystems. The study objective was to quantify the ES and disservices (DES) obtained from long-term (13-years) CA practices. Four ecosystem services (ES), i.e., provisioning, regulating, supporting, cultural services and ecosystem disservices (EDS) was assessed under maize-wheat rotation of tropical agro-ecosystems. The treatments were conventional tillage (CT), zero tillage with planting on flat land with residue (ZR + R), permanent broad beds with residue (PBB + R), permanent narrow beds with residue (PNB + R). The ES was quantified through the economic value of provisioning services, regulating services, supporting services, cultural services and EDS. The economic value of provisioning ES ranged from US$ 3105.6 ha^{-1} (CT) to US$ 3841.9 ha^{-1} (PBB + R). The CA-based practices recorded 16.2−23.7 % higher value of provisioning ES values as compared to CT. The highest economic value of regulating ES was observed under PBB + R, which was 61.1 % higher than that of CT. The economic value for SOC accumulation comprised maximum share (90.4–91.6 %) in total regulating ES value. The economic value of total ES (TES) from maize-wheat rotation was 43.0 % higher under PBB + R than that of CT. The economic value of marketed ES (MES) represents only 42.6–49.3 % of the economic value of the TES provided by maize-wheat rotation. PBB + R is a better management alternative for better TES while ensuring higher crop productivity than CT. The results can be used to formulate the payment for ecosystem scheme by the policy makers.

1. Introduction

“The ecosystem is the complex of living organisms (animal, plant, and microorganism), their physical environment and interaction among them as a system” (Millennium Ecosystem Assessment (MEA), 2005). An ecosystem service integrates functional entities such as habitats, natural biological system properties, and various ecosystem processes. These services also include ecosystems’ benefits to human populations, directly or indirectly, in goods and services. Different ecosystems offer various services that vary in quantity and quality [1]. Agricultural ecosystems, which humans design to produce essential foods, also provide other valuable by-products such as fiber and fuel, along with numerous non-marketed environmental benefits, i.e., climate regulation, carbon sequestration regulation of water flows, and water purification, maintaining genetic and biological diversity, nutrient cycling, pest regulation and pollination, recreation, scenic values, spiritual values [1–4]. Worldwide, around five billion hectares of land are used for agriculture, constituting about 38 % of the Earth’s terrestrial

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ecosystem [5]. The ES derived from farmland is heavily influenced by farmers' cultivation inputs and crop management techniques [6]. ES are typically categorized into four groups: provisioning services (food, by-product, fuel, forage, fiber, and freshwater), regulating services (regulation of climate or air quality, carbon sequestration, soil quality; flood, erosion, or disease control; and pollination), supporting services (genetic and biological diversity) and cultural services (opportunities for tourism, recreation, artistic or aesthetic appreciation, and spirituality) (MEA, 2005).

Wheat (*Triticum aestivum* L.) is a globally important crop, serving as the primary food source for the largest population after rice [7]. It is a staple for about 40 % of the world's population, providing roughly 20 % of all dietary calories and protein [8]. Similarly, maize (*Zea mays* L.), often called the "Queen of cereals," is the third most crucial cereal crop cultivated in approximately 155 countries worldwide. In India, the maize–wheat rotation (MWR) is a significant cropping system, ranking fifth in importance. It covers around 2 Mha in the Indo-Gangetic Plains (IGPs), primarily known for the rice–wheat rotation (RWR) [9]. The widespread RWR system in the northwestern IGPs has led to issues such as excessive use of natural resources, nutrient imbalances, amplified energy consumption, greater labor demands, shifts in weed populations and resistance, greater greenhouse gas emissions, and environmental degradation [10,11]. Moreover, the practice of burning rice residue poses a significant threat to the sustainability of the RWR. This method has led to severe consequences, including losing soil organic matter (SOM) and nutrients, decreased biodiversity, reduced water and energy efficiency, and deteriorated air quality. As a result, these detrimental effects have underlined the need to explore alternative maize–wheat rotation systems or adopt integrated sustainable strategies that align with the UN Sustainable Development Goals, aiming for more environmentally friendly and efficient resource use [12]. However, the traditional methods of intensive tillage and flood irrigation for maize-wheat rotation rely heavily on nitrogen fertilizers, water, and machinery. This approach has led to higher greenhouse gas emissions, a drop in groundwater levels, nitrate pollution of groundwater, and largely degradation of soil and ecosystems. Here, the major challenge is to develop the substitute production system that would be climate, resource and ecosystem resilient, and can aid to sustain the crop yields in the long-run. Recently, CA has been promoted for sustainable crop intensification [13,14]. CA, along with best management practices such as raised bed planting, offers opportunities for improving maize and wheat production in north-western IGP [15]. CA-based technologies like zero tillage, laser-aided land levelling, raised bed planting, retention of crop residue, and crop diversification have been evaluated in the IGP as alternatives to CT [16,17]. The no-till raised bed system has recently become significant in South Asia's upland cropping systems, helping to save water and mitigate the negative impacts of excess water on crops [18]. Additionally, CA practices enhance soil carbon sequestration [19] and soil health due to reduction in soil disturbance and the retention of crop residues. However, maize and wheat cultivation have significant environmental impacts, including nutrient loss (particularly N), greenhouse gases (GHGs) emissions, pesticide residues in the soil, and groundwater depletion. Despite these negative impacts, agriculture also offers positive environmental contributions through various ES, including provisioning, regulating, supporting, and cultural services. Regulating services encompass organic carbon sequestration [20,21], soil nutrient enhancement [22], pollination [23], soil retention and formation [22,24]. Supporting services include nutrient cycling [25,26], pest and disease management [27,28], water flow regulation [29], and maintaining soil fertility [30], which are essential for sustaining life on Earth. Food production, classified as a provisioning service in the Millennium Ecosystem Assessment (MEA), is vital for human survival [31] and depends on supporting and regulating services like soil fertility and pollination [32]. While agriculture is vital for providing critical provisioning services, intensive farming practices have caused the degradation of other important ES, including soil formation, soil fertility, water purification, climate regulation (Yang, 2021), and biodiversity conservation [24]. Both the positive and negative environmental impacts of agriculture can be economically valued in terms of ES and EDS.

Over the past 50 years, there has been a notable decline in global ES, with 60 % experiencing degradation [1]. Recently, the valuation of ES has gained prominence in research, not only to enhance and preserve them but also to create a basis for compensating land managers who protect and maintain them [24]. Assigning a monetary value to ES is crucial for raising awareness and highlighting its importance to policymakers [33]. In the milieu of sustainable crop production systems across several environments and enhancing production efficiency, it is crucial to evaluate all ES in different cropping system under various agroclimatic regions [24,34]Given the ongoing intensification of agricultural production, this study focuses on the ES provided by conventional and conservation management systems in maize–wheat agroecosystems. Previous research on agricultural ecosystems has led to valuations of several ES related to farming [21,22,24], but there is limited documentation specific to maize–wheat system and more specifically under different CA-based management practices. This study aims to update the scientific community on the performance of maize-wheat rotation under CT and CA practices in terms of ES. It hypothesizes that ES is declining due to current conventional agricultural management practices, and CA-based systems may offer more ES in maize–wheat agroecosystems. To verify this hypothesis, the objectives were to quantify both ES and EDS and compare these services under CT and CA–based crop management practices. The findings are intended to provide practical information to farmers, managers, and decision-makers on choosing profitable yet sustainable crop management techniques and to formulate the payment for ecosystem scheme by the policy makers.

2. Materials and methods

2.1. Experimental site

The long–term field experiment commenced in 2010 at the research farm of ICAR–Indian Agricultural Research Institute (ICAR–IARI), New Delhi, India, located at 28°35′N latitude, 77°12′E longitude, and an altitude of 229 m above sea level. The experimental site experiences a subtropical semi-arid climate characterized by dry, hot summers and cold, moist winters. The maximum temperature ranges from 40 to 46 °C in May and June, while January sees the most frigid temperatures, ranging from 5 to 8 °C. The site receives an average annual rainfall of 710 mm, with 80 % occurring during the southwest monsoon season (July to September) and the remaining amount during December and January. The average annual weather parameters during the study period are given in Fig. 1. The soil at the experimental site is classified as Typic Haplustept, with the top 0–15 cm layer exhibiting a sandy clay loam texture. Soil properties include a pH of 7.7, oxidizable soil organic carbon (SOC) content of 5.2 g kg^{-1}(Walkley & Black carbon), electrical conductivity (EC) of 0.64 dS m^{-1}, $KMnO_4$ oxidizable nitrogen (N) content of 182.3 kg ha^{-1}, 0.5 M $NaHCO_3$ extractable phosphorus (P) content of 23.3 kg ha^{-1}, and 1 N NH_4OAc extractable potassium (K) content of 250.5 kg ha^{-1}. Additionally, the soil contains adequate levels of $CaCl_2$ extractable sulfur (S) and DTPA extractable micronutrients, all above the critical deficiency thresholds.

2.2. Experimental design and agronomic management

Initially established in the 2010–11 growing season, the field experiment utilized five treatments organized in a randomized block design (RCBD) with three replications to compare conventional tillage (CT) against zero tillage on both narrow (PNB) and broad beds (PBB), with and without crop residue. From the second year onward, two additional treatments, zero tillage with residue retention (ZT + R) and zero tillage without residue (ZT), were introduced on flat land (Table 1).

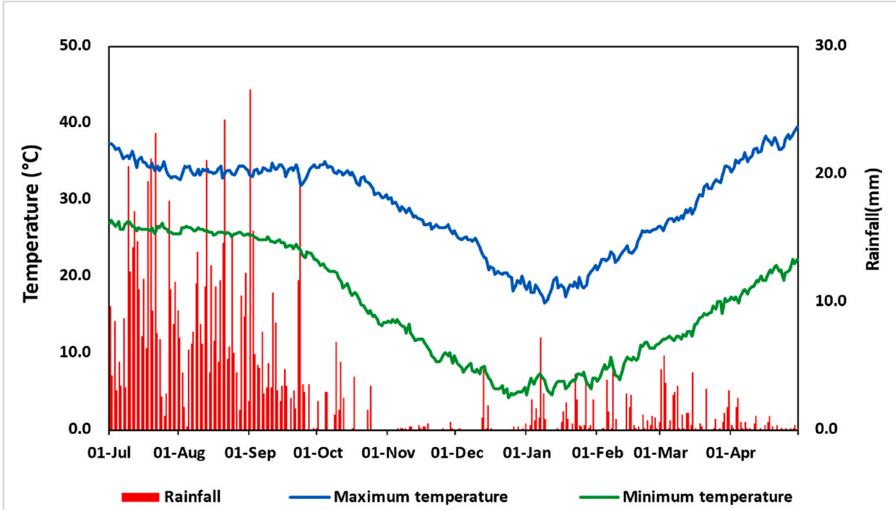

Fig. 1. Prevailing average weather condition during the crop growing seasons, 2010–2023.

Table 1
Details of treatments adopted in the experiment.

Treatment code	Type of tillage	Type of bed	Residue retention
ZT + R	Zero tillage and residue	Plain/flat land	Yes
PBB + R	Zero tillage and residue	Permanent broad bed (110 cm bed and 30 cm furrow)	Yes
PNB + R	Zero tillage and residue	Permanent broad bed (40 cm bed and 30 cm furrow)	Yes
CT	Conventional tillage	Plain/flat land	No

Each treatment was applied to a 30.0 m long and 8.4 m wide strip (approximately 252.0 m^2) to facilitate tractor operations for sowing, fertilization, harvesting, and irrigation. These strips were subdivided into three plots of 9.0 m × 8.4 m (approximately 75.6 m^2) each, serving as replications. During the initial year, the required narrow and broad bed plots were formed using a ridge/bed maker and maintained as permanent structures in subsequent years. CT plots underwent seasonal preparation involving one pass each with a tractor-drawn disk plough, cultivator, and harrow, followed by leveling to achieve a fine tilth. In contrast, no ploughing was conducted in the other plots. For PBB and PNB plots, furrows were renovated, and beds were reshaped annually before the sowing of the rainy-season maize crop. For residue removal (ZT, PPB, and PNB) and CT plots, previous crops were manually harvested at the base (3–4 cm above the soil surface), mimicking local farming practices in the IGP regions. In residue retention plots (ZT + R, PPB + R, and PNB + R), previous maize plants were manually cut 40 cm from the base and left as anchored residue. The additional loose residue was applied uniformly if needed to achieve a 40 % residue cover for each plot. Similarly, a combine harvester was used to harvest wheat plants to a height of 40 cm from the base of the plants, with the remaining crop being kept as stubble.

The maize seeds (cv. PMH 1) were sown in the first week of July 2022, grown all through the rainy season, and harvested in the third week of October 2022. Meanwhile, the wheat (cv. HDCSW 18) was sown in the second week of November 2022 and harvested during the third week of April 2023. The planting process involved various methods: a turbo seeder for plots employing PBB, PBB + R, ZT, and ZT + R techniques, a bed planter for PNB & PNB + R plots, and a seed drill for CT plots. Fertilizer boxes attached to the turbo seeder and bed planter facilitated the placement of fertilizers in the soil during sowing. A standard dose of 150 kg N, 60 kg P_2O_5, and 40 kg K_2O per hectare was administered, with total doses of P and K and half the N dose applied at sowing using the turbo seeder/bed planter (for PNB, PBB, ZT), while the remaining N was top-dressed in two equal parts (at 30 and 60 DAS in maize; after the first and second irrigation sessions in wheat). During top-dressing, fertilizers were broadcasted with careful attention, and they were applied along crop rows and on beds, avoiding furrows. Depending on rainfall and soil moisture, the maize crop received five, and the wheat crop received six irrigations at critical growth stages. Pre-sowing, glyphosate herbicide was sprayed @ 1.0 kg ha^{-1} in CA-based plots for both maize and wheat to manage existing weeds. Within two to three days of sowing, a tank mixture of atrazine at 0.75 kg ha^{-1} and pendimethalin at 0.75 kg ha^{-1} was applied to maize to control any annual weeds sprouting after the grain was planted. In wheat, isoproturon (N, N-dimethyl-N'-[4-(1-methyl ethyl)phenyl]urea) application @ 1.0 kg ha^{-1} was made 30 days after sowing for broader weed control. Notably, no fungicides or insecticides were utilized during the cultivation of wheat. However, to control stem borer and shoot flies, pesticide carbofuran 3G @ 25 kg ha^{-1} was employed in maize.

2.3. Soil sampling and analysis

Following a 13-year span of experimentation spanning from 2010 to 11 to 2022–23, soil samples were collected subsequent to the maize crop harvest. These samples were obtained from a depth of 0–15 cm in the soil, with three distinct locations sampled within each plot utilizing a core sampler. Notably, in plots subjected to PNB, two samples were retrieved from the bed areas and one from the furrow regions. Each triplicate soil sample was amalgamated to constitute a composite sample for every plot. Subsequently, these composite samples were air–dried, moved through a 2–mm sieve, and kept in plastic bags at ambient room temperature. Further processing involved finely grinding the air-dried soil samples through a 250 μm sieve, facilitating the subsequent measurement of SOC.

2.3.1. Soil bulk density

The core auger method, as described by Blake and Hartge [35], was used to determine soil bulk density (BD). The BD was then calculated by dividing the dry soil sample's weight by the core volume. This calculation was performed using the following formula (Eq. (1)):

$$BD\ \left(Mg\ m^{-3}\right) = \frac{A - B}{C} \quad (1)$$

where, A = Weight of core with oven dry soil (Mg); B=Weight of core (Mg); C=Volume of core (m^3).

2.3.2. Soil organic carbon and stock

The SOC content was estimated using the wet oxidation method described by Walkley and Black [36]. The Walkley and Black oxidizable carbon was converted to total SOC by applying a correction factor of 1.32, as suggested by Jha et al. [37]. The SOC stock (Mg ha^{-1}) for 0−15 cm soil depth was calculated following Eq. (2) [38].

$$\text{SOC stock} = \text{SOC}_{conc} \times T \times BD \times 10000 \tag{2}$$

where SOC_{conc} was expressed as kg C Mg^{-1} soil; T, depth of soil layer (m); Tadd, additional thickness (m); BD, soil bulk density (Mg m^{-3}).

To avoid bias resulting from treatment-induced BD differences (Mg m^{-3}), the 'equivalent depth basis' correction (i.e., additional thickness (T_{add}, m)) was calculated from the differences between soil mass and the equivalent soil mass and subsurface BD [39], and was added to the formula (Eq. (3)):

$$\text{SOC stock} = \text{SOC}_{conc} \times (T + T_{add}) \times BD \times 10000 \times 0.001 \tag{3}$$

2.3.3. Available soil nutrient content

The determination of available N in soil was conducted using the method developed by Subbiah and Asija [40]. Available P in soil was analyzed following the method described by Olsen et al. [41]). The method outlined by Hanway and Heidel [42] was employed to determine the available K in soil. This method quantifies the exchangeable and water-soluble K, excluding K in saline or saline-sodic soils.

2.4. Measurement of grain and stover/straw yield

The yields of maize and wheat were recorded at 12 % moisture content, while the maize stover and wheat straw yields were determined by oven–drying straw samples at 70 °C until a constant weight was achieved and expressed on a dry-weight basis. The mature maize and wheat plants were manually harvested from the net plot areas (excluding the border rows) to measure grain and stover/straw yields. The net plot area was consistent across treatments, but the number of harvested crop rows varied depending on the crop and land configuration/planting geometry. The middle four rows/narrow beds up to a 5 m length (4 × 0.7 m × 5 m = 14 m^2) from the PNB plots and the central four rows/two broad beds up to a 5 m length (14 m2) from the PBB plots were manually harvested to measure the maize yield. In wheat, four central narrow beds (2.8 m × 5.0 m) with three wheat rows per bed (totalling 12 rows) were manually harvested in the PNB plots. The PBB plots harvested two central broad beds with five wheat rows per bed (totalling ten rows). For the ZT and CT plots, 14 wheat rows were harvested from a 2.8 m × 5.0 m area. In plots with residue retention, maize crops were cut manually, and the wheat crop was harvested using a combine harvester approximately 40 cm above ground level. For residue removal and CT plots, manual harvesting was done by cutting the bases of the plants at approximately 3–4 cm above the soil. The maize grains were shelled, and wheat grains were then threshed, cleaned, and weighed to determine their yields.

2.5. Calculation and quantification of ES

In our study, we developed a comprehensive framework to quantify different ES by integrating field methods, laboratory experiments, and a questionnaire survey. The assessment of ES for maize–wheat rotation was done following the methodology outlined by Sandhu et al. [43], which involves aggregating the values of individual ES. Each ES was calculated, and the value was expressed in US dollars [44]. The components of ES measured are food, i.e., maize and wheat grain (ES1), by-products, i.e., maize stover and wheat straw (ES2), soil organic carbon accumulation (ES3), soil available N (ES4), soil available P (ES5), soil available K (ES6), water holding services (ES7), soil formation (ES8), groundwater recharge (ES9), soil fertility (ES10), biological control of pests (ES11), cultural service (ES12), greenhouse gas emission (ES13) and soil erosion (ES14). In this study, 14 services were categorized into five classes, namely provisioning services (ES1+ES2), regulating services (ES3+ES4+ES5+ES6+ES7), supporting services (ES8+ES9+ES10+ES11), cultural services (ES12) and ecosystem disservices (ES13+ES14) based on the modified Millennium Ecosystem Assessment (MEA, 2005).

Total ES (TES) values were estimated (Eq. (4)) using the formula given by Nayak et al. [24].

$$TES = ES_1 + ES_2 + ES_3 + ES_4 + ES_5 + ES_6 + ES_7 + ES_8 + ES_9 + ES_{10} + ES_{11} + ES_{12} - ES_{13} - ES_{14} \tag{4}$$

Marketed and non-marketed values of ES were assessed in the maize–wheat system. The marketed value of ecosystem services (MES) includes the total economic value derived from products such as maize and wheat grains and by-products like maize stover and wheat straw, which are directly traded by farmers in the marketplace (Eq. (5)). The remaining ES fall under the category of non-marketed ES values (NMES) (Eq. (6)) as described by Sandhu et al. [43].

$$MES = ES_1 + ES_2 \tag{5}$$

$$NMES = ES3 + ES_4 + ES_5 + ES_6 + ES_7 + ES_8 + ES_9 + ES_{10} + ES_{11} + ES_{12} – ES_{13} – ES_{14} \tag{6}$$

All services were calculated separately for the conventional and the conservation practices under maize–wheat system using the following procedures.

2.6. Provisioning services

2.6.1. Food and by-products

The economic value of food obtained from the maize and wheat crop was determined by multiplying the minimum support prices (MSP) by the grain yield, following the methodology outlined by Sandhu et al. [43]. For the fiscal year 2023–2024, the Government of India set the MSP at US$255.6 t^{-1} and US$259.9 t^{-1} to purchase food grains from farmers for maize and wheat, respectively. Additionally, the economic value of crop by-products, specifically maize stover and wheat straw, was calculated based on the farm gate prices of stover/straw bales in local markets, which was US$ 30.1 t^{-1}. The provisioning service, representing the price of raw materials produced by the maize and wheat crops, was calculated using the following Eq. (7):

$$\text{Price of raw materials}\ (\text{US\$ ha}^{-1}) = \text{Grain yield}\ (\text{t ha}^{-1}) \times \text{Price of grain}\ (\text{US\$ t}^{-1}) + \text{Straw yield}\ (\text{t ha}^{-1}) \times \text{Price of straw}\ (\text{US\$ t}^{-1}) \tag{7}$$

2.7. Regulating services

2.7.1. Soil organic carbon accumulation

One of the most advantageous aspects of CA is its potential to enhance soil carbon levels [45]. The total amount of SOC accumulated was assessed based on the equivalent market price of farmyard manure (FYM), which is valued at US$ 30.12 t^{-1}. The carbon content of FYM was measured to be 16 % by weight. The price of SOC accumulation was estimated using the following formula (Eq. (8)) given by Kumar [46].

$$\text{Price of SOC accumulation } \left(\text{US\$ ha}^{-1}\right) = \text{Amount of C accumulated in soil } \left(\text{t ha}^{-1}\right) \times \frac{100}{16} \times \text{Price of FYM} \tag{8}$$

2.7.2. Soil nutrient content

The nutrient levels in the soil, specifically N, P, and K, were quantified based on analyses of soil samples. The economic valuation of these nutrients was conducted using the replacement cost method following Eq. (9) ([47]; Kiran and Kaur, 2011). This approach involved calculating the total nutrient content (N, P, and K) and assigning an economic value based on the local market prices of the corresponding fertilizers, thus determining the monetary worth of the soil's nutrient content. Fertilizer cost is US$ 2.96 per 45 kg bag of urea, US$ 4.43 per 50 kg bag of SSP, and US$ 10.27 per 50 kg bag of MOP for available soil N, P, and K, respectively.

$$\text{Price of soil nutrient content } \left(\text{US\$ ha}^{-1}\right) = \text{Amount of available soil nutrient } \left(\text{t ha}^{-1}\right) \times \text{Price of nutrient } \left(\text{US\$ t}^{-1}\right) \tag{9}$$

2.7.3. Water holding services

Adequate availability of clean water is important for the sustainability of agro-ecosystems, with agriculture being the primary consumer, accounting for about 70 % of global water usage. Within agroecosystems, water availability depends not only on natural processes like infiltration but also on the ability of soils to hold moisture, which constitutes another crucial ecosystem service [48]. In this study's context, the irrigation water measurement employed the star-flow meter method, coupled with the determination of the wetted area of the field channel utilizing the standard flow rate equation for open channels [15]. Consequently, we measured the quantity of irrigation water saved in different CA-based management strategies compared to conventional practices. Additionally, to quantify the economic significance of the saved irrigation water (Eq. (10)), we used a cost-benefit approach, calculating the expenses associated with pumping irrigation water at a rate of US$ 0.2 per mm–ha^{-1} [15].

$$\text{Price of water holding services } \left(\text{US\$ ha}^{-1}\right) = \text{Irrigation water saved (mm)} \times \text{Expenditure to pump water } \left(\text{US\$ 0.2 mm–ha}^{-1}\right) \tag{10}$$

2.8. Supporting services

2.8.1. Soil formation

The rate at which soil forms is inseparably linked to the presence and activity of earthworms within an ecosystem. This study, assessing the ecosystem's value related to soil formation, focused on the population density of earthworms in fields. The number of earthworms was determined using the Tullgren funnel method [49]. Each earthworm possesses an average biomass of 0.2 g [50], and it is established that 1 tonne of earthworms generates 1 tonne of soil per hectare annually [43,51]. To gauge the economic worth of soil formation, the value of topsoil was calculated and multiplied by the annual soil formation rate. The estimated value of one ton of topsoil in India is US$ 1.57 t^{-1} [22], which was used to evaluate soil formation in this research (Eq. (11)).

$$\text{Price of soil formation } \left(\text{US\$ ha}^{-1}\right) = 0.2 \times 10^{-6} \times \text{Number of earthworm ha}^{-1} \times \text{Price of top soil } \left(\text{US\$ t}^{-1}\right) \tag{11}$$

2.8.2. Ground water recharge

Groundwater recharge is influenced by both rainfall and irrigation. Rainfall data was collected daily from a rain gauge at the Agro-meteorological Observatory of the Division of Agricultural Physics, IARI, approximately 400 m from the experimental plot. Measurements of irrigation water were conducted using a star-flow meter, while the wetted area of the field was determined using the standard flow rate equation for open channels [15]. Cultivating crops in bunded fields reduces runoff and increases the residence time for percolation. To estimate the economic value of groundwater recharge in bunded fields, 17 % of the total rainfall and irrigation was considered [52]. This recharged groundwater can subsequently be extracted and utilized for irrigation, thus offsetting the need to purchase irrigation water. The economic value of recharged groundwater was assessed using Eq. (12) based on the purchase price of irrigation water, set at US$ 1.5 × 10^{-3} m^{-3} water (FAO, 2004).

$$\text{Price of ground water recharge } \left(\text{US\$ ha}^{-1}\right) = \left[\text{Amount of rainfall } \left(\text{m}^3\text{ ha}^{-1}\right) + \text{Amount of irrigation applied } \left(\text{m}^3\text{ ha}^{-1}\right)\right] \times 0.17 \times \text{purchase price of irrigation water } \left(\text{US\$ m}^{-3}\text{ water}\right) \tag{12}$$

2.8.3. Soil fertility

The soil fertility service of crop fields was assessed by evaluating the soil's contribution to the crop's uptake of N, P, and K in representative soils [53]. The nutrient uptake from the soil per unit of primary produce was calculated. The contribution of nutrients from fertilizers was estimated using the standard nutrient use efficiency rates for nitrogenous, phosphatic, and potassic fertilizers in maize–wheat system, i.e., N (30 %), P_2O_5 (25 %), and K_2O (50 %) for the IGP [54,55]. The nutrient contribution of soil to crop uptake was calculated by subtracting the nutrient contribution from fertilizers from the total nutrient uptake. The economic value of soil-contributed nutrients was determined using the replacement cost approach (Eq. (13)) ([47]; Kiran and Kaur, 2011).

$$\text{Price of soil fertility } (\text{US\$ ha}^{-1}) = [\{\text{Total uptake of N by crops } (\text{t ha}^{-1}) - (\text{Amount of N applied through fertilizer } (\text{t ha}^{-1}) \times N_{\text{fertilizer}} \text{ use efficiency})\} \times \text{Price of N } (\text{US\$ t}^{-1})] + [\{\text{Total uptake of P by crops } (\text{t ha}^{-1}) - (\text{Amount of P applied through fertilizer } (\text{t ha}^{-1}) \times P_{\text{fertilizer}} \text{ use efficiency})\} \times \text{Price of P } (\text{US\$ t}^{-1})] + [\{\text{Total uptake of K by crops } (\text{t ha}^{-1}) - (\text{Amount of K applied through fertilizer } (\text{t ha}^{-1}) \times K_{\text{fertilizer}} \text{ use efficiency})\} \times \text{Price of K } (\text{US\$ t}^{-1})] \quad (13)$$

2.9. Biological control of pests

To assess the management of maize and wheat insect pests by predators and parasitoids, researchers used both natural pests and "prey surrogates" to establish a "predation rate" [56]. The predation rate in maize–wheat ecosystems was determined by evaluating the removal of prey types within a specific field based on the number of biocontrol agents present [57,58]. Sampling of both biocontrol agents and insect pests was conducted using a sweep net sampling method [59]. The economic threshold level (ETL) of insect pests, which is the point at which farmers are advised to apply pesticides, was used to estimate pest infestation per hectare at the ETL. The total pest control cost per hectare was calculated by multiplying the recommended pesticide dose with the market price and adding the application cost. The total pest control cost per hectare was used as the value of ES for biological pest control.

2.10. Cultural services

According to the MEA [1], cultural services are "the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experience, including knowledge systems, social relations, and aesthetic values." These cultural services can be assessed using the Travel Cost Method, which considers travel expenses to recreational destinations such as wildlife viewing, hunting, and fishing and the willingness to pay for outdoor educational visits. The present study divided cultural services into two categories: (1) extension of technology and (2) research. This long-term experiment, which has been ongoing since 2010, showcases various CA practices. Around 30 farmers from nearby villages, especially Pataudi village in Gurugram, Haryana, India, visited the site to learn about these CA practices during the study period. The travel cost from Gurugram to the study area is regarded as a cultural service for extending CA technologies. In the research section, two government-funded projects are associated with this experimental field, and four students have been awarded Ph.D. degrees based on their research conducted here. Due to the non-biophysical nature of research-related cultural services, their quantification and economic valuation require different techniques than those used for provisioning, regulating, and supporting services. Therefore, this study did not include research-related cultural services, but those should be considered for a more comprehensive analysis.

2.11. Ecosystem disservices

2.11.1. Green-house gases emissions

Greenhouse gas emissions negatively impact the ecosystem and must be subtracted from the TES values. The total GHG emissions (i.e., CO_2 and N_2O) during the maize–wheat crop cycles were quantified in terms of CO_2 equivalents (CO_2-eq.). As maize and wheat grow in aerobic conditions, only CO_2 and N_2O emissions were considered in the present study. Additionally, no crop biomass was burned on the farm. Established emission coefficients from the literature were used to convert field operations and applied inputs into their respective equivalent emissions [60]. This included all factors contributing to GHG emissions involved in producing and using inputs such as diesel, electricity, fertilizers, insecticides, and herbicides, expressed in units of kg CO_2-eq. ha−1. Direct N_2O emissions induced by fertilizer N application and crop residue were calculated using Eq. (14) [61,62].

$$\text{Direct } N_2O \text{ emissions } (\text{kg } CO_2 - \text{eq ha}^{-1}) = [N_{\text{fertilizer}+}\ N_{\text{residue}}]\,(\text{kg ha}^{-1}) \times 0.007 \times \frac{44}{28} \times 298 \quad (14)$$

where $N_{\text{fertlizer}}$ is the amount of N fertilizer applied (kg ha^{-1}), N_{residue} is the N contribution from crop residue [N_{residue} = Quantity of crop residue applied (kg ha^{-1}) × N content (%) of crop residue], 0.007 is the default emission factor for N fertilizer application [63], 44/28 is the conversion factor based on molecular weight of nitrogen (N_2) in relation to N_2O, 298 is the global warming potential in a 100-year horizon [64].

Volatilization and leaching losses from N fertilizer application in agricultural fields primarily contribute to indirect N_2O emissions. According to the IPCC Tier-1 guidelines, indirect N_2O emissions were calculated using modified formulas for N_2O emissions from volatilization using Eq. (15) [65] and leaching using Eq. (16) [66].

$$N_2O_{\text{volatilized}}\ (\text{kg } CO_2 - \text{eq ha}^{-1}) = N_{\text{fertilizer}}\ (\text{kg ha}^{-1}) \times 0.1 \times 0.010 \times \frac{44}{28} \times 298 \quad (15)$$

$$N_2O_{\text{leached}}\ (\text{kg } CO_2 - \text{eq ha}^{-1}) = N_{\text{fertilizer}}\ (\text{kg ha}^{-1}) \times 0.3 \times 0.0075 \times \frac{44}{28} \times 298 \quad (16)$$

where; $N_{\text{fertlizer}}$ is the amount of total N fertilizer applied, 0.1 and 0.3 are the fraction used for volatilization and leaching, respectively [67]. The coefficients of 0.010 and 0.0075 are the default emission factor used for volatilization and leaching, respectively [68].

To estimate the environmental cost of total GHG emissions from the

Table 2
Effect of CT and CA – based practices on soil bulk density, organic carbon content and organic carbon stock in the 0–15 cm soil layer.

Treatment[a]	Soil bulk density (Mg m^{-3})	Initial carbon value (g kg^{-1}) (2010)	Total organic carbon (g kg^{-1})[c]	Total soil organic carbon stock (Mg ha^{-1})
			Year- 2023	
ZT + R	1.41 ± 0.03[bc][b]	5.2	9.47 ± 0.25[a]	22.13 ± 0.71[a]
PBB + R	1.36 ± 0.02[d]		9.98 ± 0.20[a]	23.26 ± 0.13[a]
PNB + R	1.39 ± 0.03[bc]		9.29 ± 0.35[a]	21.72 ± 0.99[a]
CT	1.59 ± 0.02[a]		6.04 ± 0.04[b]	14.37 ± 0.20[b]

[a] See Table 1 for treatment details.
[b] Means followed by a similar letter within a column are not significantly different at $P < 0.05$ according to Tukey's HSD test.
[c] Soil organic carbon of the initial year(2010) was 5.2 g kg^{-1}.

maize–wheat ecosystem, the cost of one carbon emission reduction (CER) or carbon credit is used, which is approximately US$21.7 t^{-1} of carbon [69].

2.11.2. Soil erosion

Soil erosion negatively impacts the ecosystem and should be subtracted from TES values. Soil erosion was calculated from the reference data published by Ghosh et al. [20] under the maize–wheat cropping system. Notably, maize and wheat were grown in fields with bunds of 50 cm in height. Given that the average field size is under 0.1 ha, the sediment delivery ratio (SDR) is assumed to be 0.3 [70], indicating that 30 % of the eroded soil is lost from the crop fields. This ratio is applied to the reference soil erosion data specific to the region to determine the actual soil loss. Subsequently, this soil loss is converted to an economic value using the topsoil valuation of US$ 1.57 t^{-1} [22].

2.12. Statistical analysis

Soil properties, crop productivities and economic value of different ES were analyzed using ANOVA for a RCBD with three replications [71]. Tukey's honestly significant difference (HSD) test was performed as a post hoc mean separation test ($P < 0.05$) employing "agricolae" [72] package in R studio (Version 4.2.1).

3. Results and discussions

3.1. Effect of CT and CA – based management practices on soil bulk density, organic carbon content and organic carbon stock

The study found that different tillage and residue management practices significantly ($p<0.05$) affected soil BD, SOC content, and SOC stock in the 0–15 cm soil layer (Table 2). The CA-based practices showed lower BD values compared to CT. Plots with residue retention had significantly lower BD compared to those without residues. Specifically, the PBB + R treatment showed 16.9 % lower BD in the 0–15 cm soil layer than CT and was the lowest among all the treatments (Table 2). Among CA-based practices, ZT + R and PNB + R showed 12.8 % and 14.4 % lower BD than CT. There was no significant difference in BD values among the residue-treated plots, i.e., PNB + R and ZT + R. Reductions in BD as a result of CA practices have also been documented by Gathala et al. [73], Blanco-Canqui and Ruis [74], and Rao et al. [75] because of increased organic matter addition in the upper soil layer. Conventional tillage operations result through heavy machinery in physical compaction, degradation of soil aggregates, and, thereafter, a decrease in soil macrospores. The decrease in BD under CA could be due to higher SOC content, better aggregation, and increased root growth [76]. Alhameid et al. [77] reported a 23-year old notillage system in the USA that experienced lower BD than CT at all soil depths (0–7.5, 7.5–15.0, 15.0–30.0, and 30–60 cm). Even in India, after seven years of a CA-based rice-wheat system, the BD was sufficiently reduced in CA than CT [78]. Improved aggregation, higher SOM level, and increased root growth were mainly responsible for the observed reduction in BD under residue-treated plots [17].

In the present study, the highest SOC content was observed in the PPB + R treatment (9.98 g kg^{-1}), 65.2 % higher than in the CT treatment (6.04 g kg^{-1}). Similar SOC levels were noted among the other CA plots (PPB + R, PNB + R, and ZT + R), with PNB + R and ZT + R showing 53.8 and 56.89 % higher SOC content than CT. The reduction in SOC content under CT is primarily because tillage disrupts soil aggregates, increasing surface area and oxygen supply, which accelerates the decomposition of SOM [15,45]. In due course, the carbon and nitrogen in organic matter are converted to mineral forms, leading to considerable soil carbon loss [21]. The present study revealed an increase in soil organic carbon (SOC) ranging from 16.23 % under CT to 91.86 % under PBB + R compared to the SOC levels in the initial year (2010) (Table 2). Increased SOC to the extent of 3.6–6.1 % in bed planting systems, along with residue retention, has also been reported by Tripathi et al. [79]. Residues get slowly decomposed, and the resultant organic matter is added to the soil, which helps in aggregate formation and water retention and improves overall soil physical health. Removal of crop residue caused lower SOC content in CT. Moreover, tillage destroyed the stable aggregates and exposed the aggregate-protected organic C, which undergoes rapid decomposition [80,81].

In the 0–15 cm soil layer, the SOC stock followed the order: PPB + R > ZT + R > PNB + R > CT (Table 2). However, the SOC stock under PPB + R was comparable to that under PNB + R and ZT + R. CA treatments (PBB + R, PNB + R, ZT + R) contributed to soil carbon enrichment and enhanced carbon sequestration potential due to the retention of crop residues. Parihar et al. [82] found that, over five years (2008–2013), SOC concentration at a 0–15 cm depth in sandy loam soil under CA increased by 33.6–34.7 %, with an average SOC stock increase of 3.65 Mg C ha^{-1} over CT treatment. Other studies have also indicated that crop management practices like crop residue management, zero or minimal tillage, or CA can enhance soil carbon accumulation [83,84].

3.2. Effect of CT and CA–based management practices on soil available nutrients

The CA practices significantly impacted the soil's available nutrients (N, P, K) within the 0–15 cm soil layer, as shown in Table 3. The available N in the soil ranged from 226.01 to 297.63 kg ha^{-1}, with the lowest levels observed in the CT treatment and PBB + R treatment had the highest available N. The residue retention under CA reduces N mineralization and leaching losses compared to residue incorporation [85]. CA-based treatments exhibited 27.4 %–31.7 % higher available N than CT. The reduced decomposition rate of soil SOM under CA improves SOC and enriches other nutrients associated with SOM. Enhanced SOM under CA buffers nutrients like N, P, and K, replenishing plant-available pools and mitigating nutrient losses. Reduced tillage

Table 3
Effect of CT and CA–based practices on soil available nitrogen, phosphorus, and potassium in the 0–15 cm soil layer.

Treatments[a]	Available N (kg ha^{-1})[c]	Available P (kg ha^{-1})[c]	Available K (kg ha^{-1})[c]
ZT + R	288.04 ± 4.27[ab]	91.57 ± 2.63[b]	534.18 ± 14.48[b]
PBB + R	297.63 ± 3.97[a]	100.05 ± 3.54[a]	610.72 ± 10.37[a]
PNB + R	290.30 ± 4.29[a]	90.10 ± 0.85[b]	520.66 ± 13.00[b]
CT	226.01 ± 2.12b	68.58 ± 2.88[c]	421.42 ± 6.76[c]

[a] See Table 1 for treatment details.
[b] Means followed by a similar letter within a column are not significantly different at $P < 0.05$ according to Tukey's HSD test.
[c] Available N, Available P and available K of the initial year (2010) was 182.3 kg ha^{-1}, 23.3 kg ha^{-1} and 250.5 kg ha^{-1} respectively.

Table 4
Effect of CT and CA–based management practices on crop productivity.

Treatments[a]	Maize		Wheat	
	Grain yield (t ha^{-1})	Stover yield (t ha^{-1})	Grain yield (t ha^{-1})	Straw yield (t ha^{-1})
ZT + R	6.40 ± 0.25[ab]	9.87 ± 0.28[a]	5.91 ± 0.12[a]	8.19 ± 0.05[ab]
PBB + R	6.51 ± 0.25[a]	10.23 ± 0.39[a]	6.12 ± 0.02[a]	9.00 ± 0.11[a]
PNB + R	6.27 ± 0.04[a]	9.74 ± 0.11[a]	5.62 ± 0.08[a]	8.17 ± 0.06[ab]
CT	5.28 ± 0.03[b]	8.33 ± 0.04[b]	4.88 ± 0.29[b]	7.64 ± 0.07[b]

[a] See Table 1 for treatment details.
[b] Means followed by a similar letter within a column are not significantly different at $P < 0.05$ according to Tukey's HSD test.

helps stabilize soil N within soil aggregates [86]. Our findings are also supported by Dey et al. [87], who observed increased total N in CA-based rice-wheat systems compared to CT. Long-term conservation practices have significantly improved available nitrogen (N). All the treatments have shown soil increases in available N over the initial year, with 23.98 % CT, 58 % in ZT + R, 59.24 % in PNB + R, and 63.26 % in PBB + R. This substantial increase in available nitrogen highlights the potential of CA to enhance soil fertility and crop productivity. Notably, the practice of residue retention has played a crucial role in this, significantly improving the available nitrogen over non-residue plots. Reduced tillage has also been effective in stabilizing soil nitrogen within soil aggregates [86,88].

Similar to available N, the PBB + R treatment also resulted in the highest available P (100.05 kg ha^{-1}) in the 0–15 cm soil layer, which was 45.9 % greater than the CT treatment, having the lowest available P of 68.58 kg ha^{-1}. CA–based treatments had 31.4–45.9 % more available P than CT. Increased P availability may result from the release of organic acids during the decomposition process and the solubilization of native P in plots amended with residue. Moreover, the CA system with reduced tillage, which retains fresh residue, limits soil mixing of applied soluble phosphorus fertilizers. This reduces the likelihood of fixation, adsorption, and precipitation as soluble phosphate-humate complexes form, enhancing the availability of soil P [89,90]. Conversely, in CT systems, the availability of labile P decreases due to extensive soil mixing [91]. Similarly to available N, there was 194.33–329.40 % increase in available P among treatments over 13 year of conservation practices. Higher available P in CA practices has also been reported by Iqbal et al. [92], Bhattacharya et al. [93] and Das et al. [94]. It was also reported that residue retention in plots had improved the soil available P. This improvement can be attributed to the release of organic acids during the decomposition of organic matter, a process that favours the solubilization of native P in residue retention plots. Furthermore, enhanced SOM with fresh residue retention under a CA system with reduced tillage ensures minimal soil mixing of the applied fertilizer soluble P. This reduction in soil mixing provides reassurance about the effectiveness of CA practices, leading to lesser chances of fixation, adsorption and followed by precipitation, by the formation of soluble phosphate-humate complexes, which enhance the liability and availability of soil P [89,90,95,96].

For available K, the levels in the 0–15 cm soil layer followed the order: PBB + R > ZT + R > PNB + R > CT, with values ranging from 421.42 to 610.72 kg ha^{-1}. The PBB + R treatment had significantly higher available K, 44.9 % more than the CT treatment. In CA practices, cereal residues contribute more K to the soil due to their higher K content in the biomass (Meean et al., 2018). Parihar et al. [97] also observed that the CA system enhances the availability of soil N, P, and K due to improved soil aggregation. This better aggregation helps protect SOM and the nutrients associated with it, reducing nutrient losses from the soil. Long term Conservation agriculture (13 years) has shown significant improvement in soil available K. In CT, ZT + R, PBB + R, and PNB + R, an increase of 68.23, 113.25, 143.80, and 107.85 % in soil available K over the start of the experiment was found, respectively.

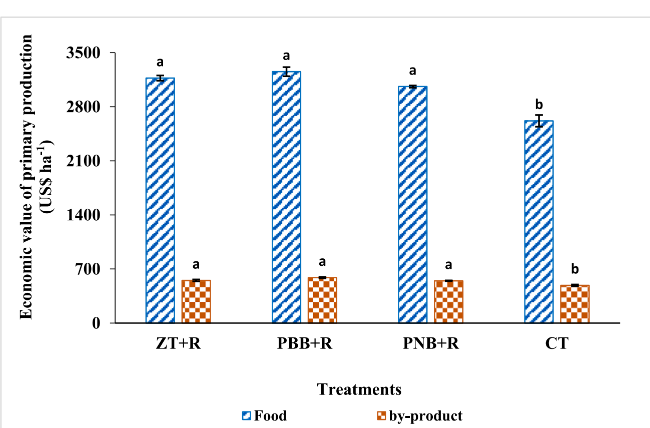

Fig. 2. Effect of CT and CA–based management practices on economic valuation of provisioning ES.

3.3. Effect of CT and CA–based management practices on grain and stover/straw yield

Crop establishment practices had a statistically significant impact ($p<0.05$) on both the grain and stover/straw yields of maize and wheat (Table 4). Practices based on CA improved grain yields by 18.8–23.3 % and 15.2–25.4 % for maize and wheat, respectively, than the CT, which is the traditional farmers' practice. Similarly, the stover yield of maize and straw yield of wheat increased by 16.9–22.8 % and 6.9–17.8 %, respectively, under CA-based practices compared to CT. Among the management practices, the PPB + R method significantly outperformed others, increasing grain yields of maize and wheat by 1.23 t ha^{-1} and 1.24 t ha^{-1}, respectively, compared to CT. These findings align with previous research in IGP, which reported higher crop yields under CA compared to conventional tillage (CT) in maize–wheat systems [11,17,98]. The higher crop yields associated with CA can be ascribed to multiple factors: the addition of nutrients [99], decreased weed populations [100], better soil physical condition [11,101], improved water management practices [17], and greater nutrient use efficiency [102] in comparison to CT. However, the PBB + R treatments resulted in significantly higher productivity than CT [103]. In the case of maize, two rows of corn planted on the edges of the beds or close to the furrows under PBB + R might not have been subjected to water stress. If any tension is seen at all, further sporadic rainfall may help. Moreover, the even distribution of residue on top of the broad beds in PBB + R improved water infiltration and conservation, reduced runoff and erosion, moderated temperature, controlled weeds, and increased soil microbial activity, leading to biological tillage [15].

Further research on photosynthesis, light interception, root water uptake, nutrient load, and radiation-use efficiency in relation to crop geometry under different bed configurations would provide deeper insights into crop productivity [16]. In the current study, three rows of wheat were sown at approximately 14 cm spacing on each narrow bed, potentially causing overcrowding and reduced tillering [104]. However, this arrangement was more effective in smothering weeds [100].

3.4. Provisioning ecosystem services

The provisioning ES, which includes food and by-products, was calculated based on the MSP for maize and wheat grain and local market values for maize stover/wheat straw, respectively. The result showed that PBB + R had the highest (US$ 3253.87 ha^{-1}) and CT the lowest (US $ 2617.28 ha^{-1}) food ES values (Fig. 2). CA-based management practices resulted in higher (17.0–24.3 %) food ES values compared to the CT. Food ES values under PBB + R, PNB + R, and ZT + R were 24.3 %, 17.0 %, and 21.2 % higher over the CT though PBB + R found at par with ZT + R and PNB + R. For by-product values, PBB + R registered the highest value (US$ 588.04 ha^{-1}), which was 20.4 % higher than CT, but it was statistically similar to ZR + R and PNB + R (Fig. 2). The total provisioning ES value for food and by-products ranged from US$ 3105.61 ha^{-1} (CT) to US$ 3841.91 ha^{-1} (PBB + R) with a mean value of US$ 3570.02 ha^{-1}(Fig. 4). The PBB + R treatment resulted in highest crop yields (6.51 and 6.12 t ha^{-1} for maize and wheat grain and 10.23 and 9.00 t ha^{-1} for maize stover and wheat straw, respectively) compared to conventional tillage (CT), which yielded lowest (5.28 and 4.88 t ha^{-1} for maize and wheat grain and 8.33 and 7.64 t ha^{-1} for maize stover and wheat straw, respectively). The highest yield contributed to the highest valuation of provisioning ES values under the PBB + R treatment. Results also revealed that the CA-based practices recorded 16.2–23.7 % higher total provisioning ES values than CT. Pathak et al.

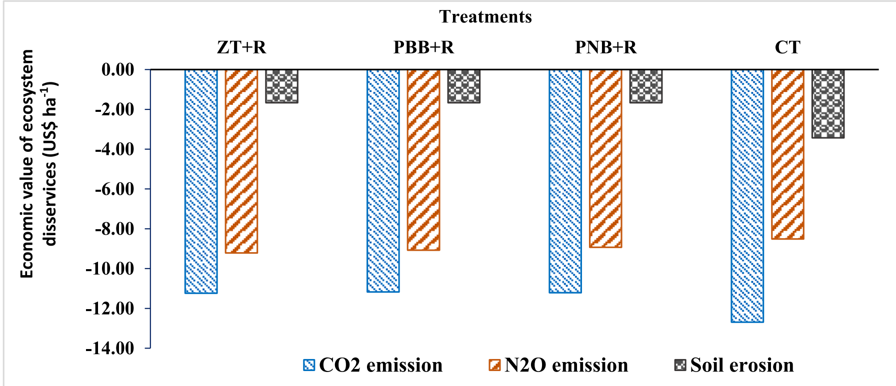

Fig. 3. Effect of CT and CA–based management practices on economic valuation of ecosystem disservices.

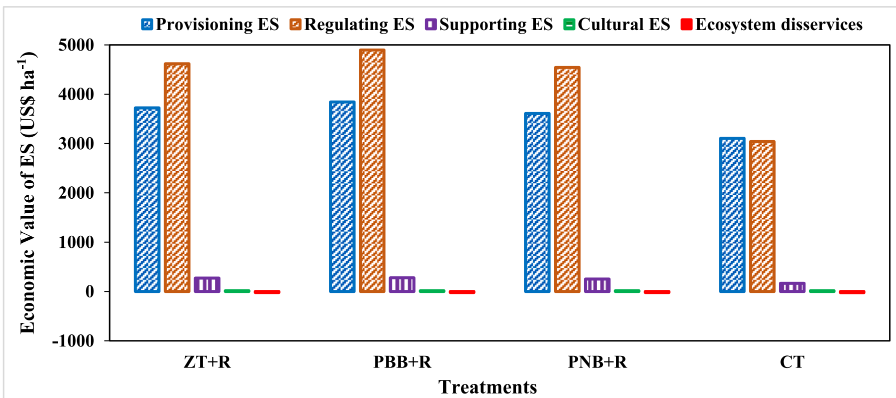

Fig. 4. Effect of CT and CA–based management practices on economic valuation of different ES.

Table 5
Effect of CT and CA–based management practices on economic valuation of different regulating ES.

Treatments[†]	SOC stock (US$ ha^{-1})	Available soil N (US$ ha^{-1})	Available soil P (US$ ha^{-1})	Available soil K (US$ ha^{-1})	Water holding service (US$ ha^{-1})	Regulating ES (US$ ha^{-1})
ZT + R	4228.52 ± 136.48[a]*	41.19 ± 0.61[a]	116.05 ± 3.33[b]	219.52 ± 1.35[b]	11.21 ± 5.95c	4616.48 ± 140.28[a]
PBB + R	4445.52 ± 25.39[a]	42.56 ± 0.57[a]	126.80 ± 4.49[a]	250.97 ± 4.26[a]	31.60 ± 1.17[a]	4897.45 ± 16.95[a]
PNB + R	4151.56 ± 189.13[a]	41.51 ± 0.61[a]	114.18 ± 1.08[b]	213.96 ± 5.34[b]	20.38 ± 2.33[b]	4541.60 ± 182.24[a]
CT	2747.09 ± 37.29[b]	32.32 ± 0.30[b]	86.91 ± 3.65[c]	173.18 ± 2.78[c]	0.00 ± 0.00[d]	3039.50 ± 38.79[b]

[22] reported that the value of raw materials produced under wheat cultivation was lower (US$ 806 ha^{-1}) under CT and higher (US$ 851 ha^{-1}) under ZT. In the eastern region of India, Nayak et al. [24] found that the market value of ES, including grain and straw from rice farms across four agro-climatic zones, ranged from US$ 1052 to 1234 ha^{-1} y^{-1}, with an average of US$ 1122 ha^{-1} y^{-1}. Yu et al. [105] investigated that the economic value of provisioning service provided by ten typical rice paddies in China ranged between $ 1484 ha^{-1} yr^{-1} to $ 3564 ha^{-1} yr^{-1}. Sandhu et al. [43] reported a higher economic value of provisioning service, *i.e.*, food under organic fields ($ 3990 ha^{-1} yr^{-1}) as compared to conventional fields ($ 3220 ha^{-1} yr^{-1}) in the Canterbury region of New Zealand, highlighting the intriguing potential for economic benefits in different farming practices.

3.5. Regulating ecosystem services

3.5.1. Soil organic carbon accumulation

The soil analysis results indicated a significant effect ($p < 0.05$) of various tillage, residue, and management practices on ES value for SOC accumulation. The economic value of SOC accumulation was highest in the PPB + R treatment at US$ 4445.52 ha^{-1} and lowest in the CT treatment at US$ 2747.09 ha^{-1} (Table 5). Yu et al. [105] estimated ES values of SOC accumulation ranging from US$ 494–3487 ha^{-1} in ten rice varieties in China. Similarly, Lv et al. (2008) estimated the ES value due to SOC accumulation at US$ 2.28 × 10^9 annually from rice-wheat farming in China's lower Yangtze River region. Sandhu et al. [106] evaluated the economic value of SOC accumulation based on carbon emission reduction (CER) or carbon credits, reporting estimates of US$ 22 ha^{-1} y^{-1} for organic fields and US$ 20 ha^{-1} y^{-1} for conventional fields. In the present study, the economic value of SOC accumulation under CA-based practices was 51.1–61.8 % higher than under CT. This may be attributed to the increased SOC content resulting from the lack of tillage and the retention of crop residues [22]. Palm et al. [107] demonstrated that CA alters soil properties and processes compared to CT systems, impacting ES, including climate regulation through carbon sequestration and GHG emissions reduction. The monetary value of carbon sequestration varies depending on the region and management practices, but on average, it has been estimated at around $ 50 per ton of

CO_2 equivalent sequestered [108,109]. The monetary value of carbon sequestration in the hilly agroecosystem of Manipur, under a long-term study on rice-based cropping systems, was estimated to range between $ 30 and $ 70 per ton of CO_2 equivalent, based on the observed carbon accumulation rates [110].

3.5.2. Soil available nutrients

The study found that CA-based practices significantly impacted soil nutrient availability (N, P, and K). The economic value of soil available N ranged from $32.32 to $42.56 ha^{-1}, with the highest value observed in the PBB + R treatment, which is 31.7 % higher than the CT (Table 5). However, the residue retention treatments (PBB + R, PNB + R, ZT + R) showed similar economic values for soil available N. Results revealed that CA practices resulted in 27.4–31.7 % higher economic value for soil available N than the CT. Enhanced SOM under CA buffers nutrients like N, P, and K, replenishing plant-available pools and mitigating nutrient losses. Reduced tillage helps stabilize soil N within soil aggregates [86].

In the case of soil available P, the economic value of ES followed the order: PBB + R > ZT + R > PNB + R > CT (Table 5), ranging from US$ 86.91 to 126.80 ha^{-1}. The PBB + R treatment produced the highest value for available soil P, which was significantly 45.9 % higher than that of CT. Minimal soil disturbance under CA reduces P fixation, while crop residues left on the surface decompose over time, releasing organic P that matches crop needs [90,111].

Like available N and P, the highest ES value for soil available K was also observed in the PBB + R treatment, US$ 250.97 ha^{-1}, while the lowest was in the CT treatment, US$ 173.18 ha^{-1} (Table 5). CA–based plots exhibited 23.5–44.9 % higher ES values for available K compared to the CT plot. Cereal residues, particularly from wheat, are rich in K and, when retained, significantly add K to the soil [97]. Higher residue retention in CA practices (PBB + R, PNB + R, ZT + R) also contributes to non-exchangeable K in the soil, enhancing the soil's buffering capacity and K bioavailability (Meena et al., 2018).

Our findings indicated that CA–practices led to significantly 26.4–43.7 % higher ES values for all soil macronutrients than the CT. The absence of tillage and consistent residue retention increased SOM and enhanced the physical and biological properties of the soil. These improvements are likely responsible for the highest nutrient-based ES value observed under permanent bed planting with residue retention (PBB + R), estimated at US$ 420.33 ha^{-1} (Table 5). CA systems improve the availability of N, P, and K through better soil aggregation, protecting SOM and associated nutrients within soil aggregates and reducing nutrient losses [97]. Similarly, using the market price of chemical fertilizers, the annual value of biological nitrogen fixation services for rice was approximately US$ 13.65 ha^{-1} y^{-1}, as Santos et al. [112] reported from the Ecological Station of Jatai.

3.5.3. Water holding service

In CA-based systems, higher soil moisture content led to an increased water-holding capacity compared to CT treatments. Since the CT plots had the lowest soil moisture content, they were used as a reference to measure the water saved by other treatments. The water holding capacity for the CT treatment was set to zero as a baseline. The highest economic value of water holding services was observed under PBB + R, amounting to US$ 31.60 ha^{-1} (Table 5). This improvement is primarily due to reduced evaporation loss from the soil, as crop residues cover the soil surface, leading to decreased irrigation water usage in residue-retained plots [15,17]. Pathak et al. [22] also reported that zero tillage and zero tillage with residue retention treatments resulted in higher soil moisture content and, consequently, higher water-holding service values than conventional treatments. These findings have significant practical implications, particularly in the Indo Gangetic Plains (IGP), where ZT saves irrigation water in the range of 20–35 % in the wheat crop compared to CT, with reduced water use by about 10 cm ha^{-1} [113]. Water productivity increased in residue retention CA plots over non-residue and CT plots in a cotton-based system [16]. This significant increase in water productivity is primarily attributed to the role of residue retention in the CA system. The residual moisture, which would otherwise be lost to evaporation, is productively utilized by the succeeding crop, thereby enhancing water productivity. This understanding of the key factors influencing water productivity can guide future sustainable farming practices.

The highest value of regulating ES was observed under PBB + R (US$ 4897.45 ha^{-1}), and the lowest value under CT (US$ 3039.50 ha^{-1}) (Table 5), which was 61.1 % lower than that of CT. The CA−based practices showed a 49.4−61.1 % higher value of regulating ES than the CT. Out of total regulating ES, the economic value for SOC accumulation comprised maximum share (90.4−91.6 %), followed by available−K (4.7−5.7 %), available−N (0.9−1.1 %), available−P (2.5−3.0 %) and water holding services (0.2−0.6 %), respectively. Among the CA−based treatments, those retaining residue exhibited significantly greater regulating ES values (31.8 %) than those without residue retention.

3.6. Supporting ecosystem services

3.6.1. Soil formation

The economic value of soil formation, influenced by earthworm populations, varied significantly across different treatments (Table 6). This value ranged from US$ 0.29 ha^{-1} y^{-1} in the PBB + R treatment to US$ 0.08 ha^{-1} y^{-1} in the CT treatment, with an average value of US$ 0.23 ha^{-1} y^{-1}, which includes only the activity of earthworms; other drivers of soil formation have not been accounted for in this study. Earthworms are crucial in forming soil and offering ES (Wright and Jones, 2003). Often referred to as "nature's plough" and "ecosystem engineers" (Hale et al., 2008), earthworms significantly contribute to soil health and ecosystem functioning. They thrive better in moderately humid soils than in dry upland soils (Bremen and Buurman, 2002). A significant 221.3.3−266.7 % improvement in the value of soil formation has been observed under CA-based practices as compared to the CT. The absence of tillage in CA plots likely reduced direct physical damage to earthworms and their habitat [114]. Briones and Schmidt [115] also conducted a global meta-analysis. They discovered that practices involving minimal soil disturbance, such as no-tillage and conservation agriculture, significantly boosted earthworm abundance (137 and 127 %, respectively) and biomass (196 and 101 %, respectively) compared to conventional ploughing. The retention of crop residues increased the organic matter in the soil, providing a primary food source for many earthworm species and thereby boosting their population [114]. Similarly, Pathak et al. [22] found that the value of soil formation due to earthworms was highest under ZT + R, followed by ZT and CT in wheat

Table 6
Effect of CT and CA – based management practices on economic valuation of different supporting ES.

Treatment[a]	Soil formation (US$ ha^{-1})	Ground water recharge (US$ ha^{-1})	Soil fertility (US$ ha^{-1})	Biological control of pest (US$ ha^{-1})	Supporting ES (US$ ha^{-1})
ZT + R	0.29 ± 0.02[ab]	3.73[b]	252.09 ± 6.66[ab]	14.80	270.90 ± 6.69[ab]
PBB + R	0.29 ± 0.02[a]	3.48[d]	259.13 ± 2.74[a]	14.80	277.70 ± 2.76[a]
PNB + R	0.26 ± 0.02[a]	3.61[c]	234.26 ± 1.23[b]	14.80	252.93 ± 1.23[b]
CT	0.08 ± 0.01[b]	3.86[a]	148.33 ± 5.09[c]	14.80	167.07 ± 5.11[c]

[a] See Table 1 for treatment details.
[b] Means followed by a similar letter within a column are not significantly different at $P < 0.05$ according to Tukey's HSD test.

production. The value of biodiversity and soil formation for future generations underscores the need for understanding the spatial and temporal distribution of these benefits. This understanding is crucial for effective resource mobilization and management, a responsibility that we all share.

3.6.2. Recharge of ground water

Groundwater recharge is crucial in maintaining ecosystem water supply by stabilizing and regulating the hydrological cycle. This includes water infiltration into soils and aquifers, moderation of runoff, and plant transpiration [116]. The estimation of water contribution to groundwater recharge varied depending on topographic conditions, irrigation water volume, and precipitation levels. This study considered water inputs such as rainfall and irrigation data for each treatment to calculate the economic value of this service in agricultural fields. In the study area, both maize and wheat crops are grown under irrigated. The mean economic value of ES was determined to be US\$ 3.67 ha^{-1}, attributed to groundwater recharge across all plots. Notably, the CT plot demonstrated a slightly higher value at US\$ 3.86 ha^{-1}, while the PBB + R plot exhibited a lower value at US\$ 3.48 ha^{-1} under the maize–wheat system (Table 6). This variation is attributed to the differing amounts of irrigation water applied. Additionally, the climatic condition of the study area is semiarid, which experienced a lower amount of rainfall (~87 cm) during the cropping season, leading to a relatively lower average economic value for groundwater recharge during crop cultivation. Nayak et al. [24] reported a monetary value of US\$ 11–12 $ha^{-1}y^{-1}$ for hydrological flow services in rice fields. Similarly, Porter et al. [117] estimated the economic value of hydrological flow services to be US\$ 86 ha^{-1} y^{-1} for cereals and US\$ 76 $ha^{-1}y^{-1}$ for pastures. New Zealand, provide \$ 54 t ha^{-1} yr^{-1} for hydrological flow, while in Taastrup, Denmark, cereals and pastures contribute \$ 86 and \$ 76 t ha^{-1} yr^{--1}, respectively [43,118]. However, the most promising aspect is the potential of CA practices, which clearly show massive potential in sustaining groundwater levels. This data brings hope for the future of sustainable agricultural practices.

3.6.3. Soil fertility

Soil fertility is crucial for sustainable agricultural production. Effective agronomic practices, including organic farming, reduced tillage, crop rotation, residue retention, legume cultivation, and chemical fertilization, can help maintain or enhance soil fertility and boost grain yields [119,120]. Significant variations in ES values attributable to soil fertility status were observed across CA and CT, with values ranging from US\$ 148.33 ha^{-1} (CT) to US\$ 259.13 ha^{-1} (PBB + R) (Table 6). Previous studies have estimated ES related to soil fertility at US\$ 281 ha^{-1} for cultivated land [121] and US\$ 281 ha^{-1} for rice-based systems [24]. In the present study, the CA–based treatments exhibited a 57.9.9−74.7 % higher value of ES for soil fertility than the CT. During harvest, crop residue remains as stubble in residue-retained plots, whether harvested manually or mechanically. Maize stover typically contains about ~1.1 % N, ~0.34 % P, and ~1.58 % K; wheat straw contains about ~0.44 % N, ~0.05 % P, and ~1.64 % K [122]. Over 13 years, plots with residue retention, such as PNB + R, PBB + R, and ZT + R, significantly increased the inputs of N, P, and K from the residue compared CT treatment, leading to notable improvements in soil fertility. During harvest, crop residue remains as stubble in residue retention plots, whether harvested manually or mechanically. Over 13 years, plots with residue retention, such as PNB + R, PBB + R, and ZT + R, impressively increased the inputs of N, P, and K from the residue compared to CT treatment, leading to significant improvements in soil fertility.

3.6.4. Biological control of pest

Maize and wheat crops are vulnerable to numerous insect pests, with aphids posing a significant threat to wheat yield in the studied area. Specifically, two types of wheat aphids, *Rhopalosiphum padi* L. and *Sitobion avenae* F., were observed infesting wheat fields. The presence of predatory coccinellid beetles, including the six-spotted ladybird beetle (*Cheilomenes sexmaculata* Fab.) and the seven-spotted ladybird beetle (*Coccinella septempunctata* L.), was noted, as they were seen preying on the wheat aphids. Consequently, these beetles were recognized as effective biological control agents for these pests. The economic value of biological pest control services was US\$ 14.80 ha^{-1} across all treatments (Table 6). This value remained consistent across treatments due to minimal variation in wheat aphids and coccinellid beetle populations among the different experimental conditions. Coccinellid beetles are significant agricultural predators due to their aggressive feeding behaviour and have been employed as biological control agents against various sap-sucking pests. The population dynamics of natural enemies generally follow those of their prey, maintaining a balance between pest and predator populations [123]. The presence of these predators can reduce the need for excessive pesticide use, thereby promoting the conservation and effectiveness of natural enemies and providing economic benefits to farmers [124]. Numerous studies have highlighted the importance of predator abundance and diversity for successful pest control [125]. Nayak et al. [24] reported an average economic value of US\$1.6 $ha^{-1}year^{-1}$ as control of pest services in rice cultivation. However, intensive pesticide use can hinder the ability of agricultural systems to support natural pest control services [57].

The comprehensive value of supporting ES varied between US\$ 167.07 ha^{-1} for CT and US\$ 277.70 ha^{-1} for PBB + R, with an average value of US\$ 242.15 ha^{-1} (Table 6). The highest value of supporting ES was under PBB + R, similar to the ZT + R. The findings indicate that CA-based practices yielded supporting ES values 23.8−66.2 % higher than CT's. Within the CA-based treatments, retaining residue exhibited significantly greater total supporting ES values (20.4−26.1 %) than those without residue retention. Out of total regulating ES, the economic value for soil fertility comprised the maximum share (88.8−93.3 %), followed by biological pest control (5.3−8.9 %), groundwater recharge (1.3−2.3 %), and soil formation (0.1 %), respectively.

3.7. Cultural ecosystem services

The current investigation delineates cultural services into two categories: (1) technology extension and (2) research. The evaluation of cultural services is exclusively considered to be worth attributed to technology extension. Due to its non-biophysical and intangible nature, the quantification and economic assessment of research-linked cultural services were not conducted (Milchu et al., 2013; [21,126]). In the present study, the travel cost from Gurugram to the study area is regarded as a cultural service for extending CA technologies. It was found that the estimated value of cultural services amounted to US\$ 15.72 ha^{-1} for all treatments (Fig. 4). The maize–wheat cropping system in the Indo-Gangetic Plains (IGP) has traditionally relied on extensive conventional tillage, resulting in limited information on conservation practices in this area. Our Institution, a pioneer in the country, has been involved in many extensions and farmers' training programs country-wide. The research field has been under CA-based practices for the last 13 years. It plays a crucial role in providing cultural services through technology transfer to the agricultural community in the IGP region of South Asia. Some farmers, agricultural experts, and students visited long–term conservation field experiments during the growing season. Industrialized cultures often prioritize cultural ES over other services, recognizing their significance in various contexts [3,127]. But even with this acknowledged significance, the use of CES in decision-making lags well behind that of more tangible services [128,129]. This is mainly because accurately assessing and quantifying cultural services is fraught with many challenges [130]. The practical application of the research was crucial, given the non-biophysical and intangible nature of the research-linked cultural services, which made their quantification and economic assessment challenging ([131]; Milchu et al., 2013; [21,126]). Despite their crucial role in human well-being, their intangible nature presents challenges for quantification and integration into economic evaluations and landscape planning [131, 132]. The lack of spatially explicit data may be a significant reason for excluding a broader range of cultural services in these studies [133, 134].

3.8. Ecosystem disservices

3.8.1. Greenhouse gases emissions

The cultivation of maize and wheat has resulted in various environmental externalities, such as the release of GHGs, such as carbon dioxide (CO_2), primarily from the combustion of fossil fuels in farm machinery and farm inputs, as well as nitrogen loss through leaching and volatilization. GHG emissions are commonly quantified in terms of carbon dioxide equivalent (CO_2-eq) per hectare (t ha^{-1}), with their economic impact assessed based on the trading value of one carbon credit, estimated at approximately US$ −21.7 t^{-1} of carbon [69]. Results indicated that the highest environmental cost for CO_2 emission under CT was estimated at US$ −12.69 ha^{-1}, whereas treatment like PBB + R showed the lowest costs at US$ −11.18 ha^{-1} (Fig. 3). CA−based practices resulted in around 13 % lower environmental costs for CO_2 emission as compared to the CT. The fewer field operations using farm machinery caused lower fossil fuel consumption, resulting in lower CO_2 emissions under CA compared to the CT system [62,135,136]. The environmental cost for N_2O emission through N leaching and N_2O volatilization ranges from US$ −8.51 ha^{-1} to US$ −9.21 ha^{-1}(Fig. 3). Plots under residue retention resulted in 4.7–7.6 % higher N_2O emission based environmental cost as compared to residue removal plots. This may be attributed to the additional N_2O emissions from retained crop residues. Bhatia et al. [137] also reported higher N_2O emissions under ZT + R than in ZT and CT systems. The highest negative ES value for GHG emissions was observed in the CT treatment (US$ −21.20 ha^{-1}), while the lowest negative value was found in the PBB + R treatment (US $ −19.71 ha^{-1}). The average ES value for GHG emissions across all treatments was calculated as US$ −20.51 ha^{-1}. CA systems demonstrated a substantial reduction (3.7−7.5 %) in environmental costs related to total GHG emissions compared to CT systems. Pathak et al. [22] reported similar findings, where the environmental cost for CT wheat was US$ −12.93 ha^{-1}, compared to US$ −12.25 ha^{-1} for ZT wheat and US$ −12.17 ha^{-1} for ZT + R. This indicates that adopting ZT and shifting from CT-based to CA-based wheat production could be an effective strategy for reducing the GHG footprint of wheat production [138].

3.8.2. Soil erosion

Soil loss through erosion is one of the significant disservices to the environment. Based on soil erosion data collected from literature published by Ghosh et al. [20], the net economic values of soil loss in all the treatments were calculated. Results showed that the soil erosion ES value varied from US$ −3.43 ha^{-1} to −1.67 ha^{-1} (Fig. 3). Bunded crop plots modified the sediment delivery ratio and reduced the soil loss, resulting in a lower negative economic value of soil erosion. A study conducted on the rice system found that the environmental cost due to soil erosion was valued at US$ −4.2 to −2.1 ha^{-1} y^{-1} [24]. Another study conducted in Japan found that the environmental function of agriculture concerning erosion control was valued at US$ 4147 million over the whole country [139]. In the present study, CA-based practices resulted in 25.7–51.4 % lower soil erosion-based environmental costs than the CT. This may be attributed to crop residue mulching, which protects soil from the impact of raindrops [20]. Additionally, crop residues enhanced SOC and improved soil structure, leading to higher water penetration into the soil and preventing runoff and splash erosion [140]. Delgado et al. [141] also reported that effective management and applying robust conservation techniques, including the return of crop residues to the soil, minimal tillage, and cover crops, play a crucial role in maintaining soil cover and reducing soil erosion. This knowledge of specific practices empowers us in our efforts for soil erosion control. Globally, the monetary value of soil erosion prevention in rice production systems is estimated to range from $ 4 to 327 million yr^{-1}, with a specific value of $ 147 million yr^{-1} for India [142].

The comprehensive value of EDS varied between US$ −24.64 ha^{-1} and US$ −21.81 ha^{-1}, with an average value of US$ −22.62 ha^{-1} (Fig. 4). The findings indicate that CA–based practices resulted in a 11.4−13.0 % lower value for EDS than those of CT. Out of total EDS, the environmental cost for CO_2 emission comprised the maximum share (50.8−51.5 %), followed by N_2O emission (34.6−41.7 %) and soil erosion (7.5−13.9 %), respectively.

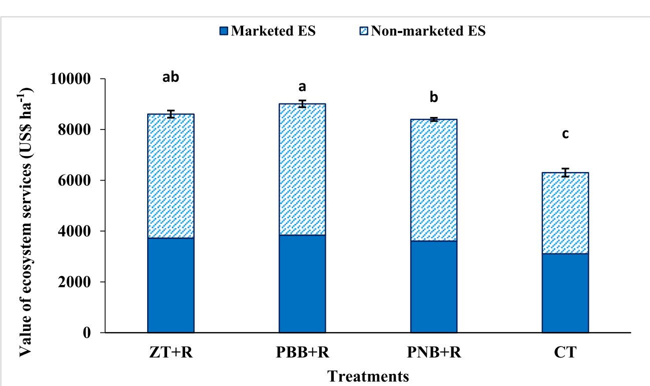

Fig. 5. Effect of CT and CA–based management practices on marketed and non-marketed ES.

3.9. Marketed and non-marketed ES

The total ecosystem services (TES) from the–wheat cropping system were significantly ($p<0.05$) affected by different tillage, residue, and crop establishment practices (Fig. 5). The TES under various management practices ranged from US$ 6303.27 ha^{-1} to US$ 9010.86 ha^{-1} with a mean value of US$ 8079.04 ha^{-1} (Fig. 5). The CA-based practices resulted in a 33.2–43 % increase in TES as compared to the CT. The PBB + R was superior among all the treatments tested. PBB + R plots resulted in US$ 2707.59 ha^{-1} (43 %) higher in TES values over CT plots. This supports our hypothesis that CA practices, especially PBB + R, improve primary production, enhance supporting and regulating services, and reduce environmental externalities, thereby delivering greater ES compared to CT in the maize–wheat system. Pathak et al. [22] also found a higher TES value under zero tillage with residue retention than CT. Moushani et al. [21] also reported a higher ES in soybean cultivation under CA than CT.

Till now, the total economic value of the maize–wheat cropping system in the IGP region has been assessed primarily through its direct outputs (i.e., MES), namely grain and stover/straw, within socio-economic statistical systems, but the remaining ES (i.e., NMES) is never considered as a part of general accounting and remains outside economic decision making. The approach used here demonstrates the value of NMES in addition to the usual market value of ES in the maize–wheat system. The result indicated that the MES represented only 42.6–49.3 % of the economic value of TES provided by the maize–wheat system (Fig. 5). The remaining 50.7–57.4 % comes from a variety of NMES, including regulating services (SOC accumulation, soil nutrient availability, and water holding service), supporting services (soil formation, groundwater recharge, soil fertility, biological pest control) and cultural services (technology extension), and EDS (including GHG emissions and soil erosion). Wang et al. [143] estimated the total economic value of ES to be US$ 10807 ha^{-1} y^{-1}, which includes primary production, social security, gas regulation, SOC accumulation, and water conservation. Notably, they found that 74 % of this value was attributable to ES, excluding primary production. Similarly, Qin et al.

[144] calculated the economic value of various ES in conventional and rice-duck paddy systems to be between US$ 2236 and US$ 2650 ha^{-1} y^{-1}, with ES (excluding primary production) comprising 63 % of the total value. In contrast, Nayak et al. [24] reported that the economic value of rice-related ES, including food production, by-products, pest biocontrol, soil formation, nutrient mineralization, carbon flow, nitrogen fixation, soil fertility, hydrological flow, and soil erosion, ranged from US$ 1238 to US$ 1688 ha^{-1} y^{-1}. ES, other than primary production, accounted for only 11–34 % of the total economic value. In a rice-wheat system in North India, Pathak et al. [22] found that the provisioning value of wheat alone was US$ 1035 ha^{-1}, making up 93.6–97.6 % of the total economic value when considering soil nutrient enrichment, soil carbon addition, water retention, and soil formation.

In the present study, the economic value of NMES amounted to a higher share of the TES than the MES. These NMES are the 'shadow prices' of ES, which are generally not exchanged in markets but are traded against each other in agricultural landscapes (Sindhu et al., 2008). The present work also indicates that CT resulted in a decline in some of these services compared with CA, with an associated lower economic value for NMES in CT by 14.3–61.6 % than CA-based practices. The current work put forward a new approach to looking at the future of maize–wheat cropping system by considering ES as an essential factor in production and indicates that it should be included in decisions and policy-making by different stakeholders [145].

4. Conclusion

This study proposes a framework to quantify the economic valuation of marketable (provisioning ES) and non-marketable ES (regulating ES, supporting ES, cultural ES, and EDS) from the maize–wheat system. This study also investigated the complex interplay between different tillage, residue, and crop establishment practices and ES, emphasizing and evaluating the critical importance of agroecosystem management in influencing various ES indicators. The result indicated that plots under permanent broad bed and residue retention (PBB + R) had about 23.7 % higher provisioning ES than CT (farmers' practice) plots, and PBB + R had significantly higher provisioning ES than other promising and novel CA practices like PNB + R and ZT + R. The CA–based systems with more accumulation of SOC and enhancement of available N, P, K, and water holding capacity led to significant improvement in regulating ES value by 49.4–61.1 % than CT. Similar was the case for supporting ES, where it was 51.4–66.2 % higher under the CA-based practices than the CT. The CA–based systems with more accumulation of crop residues on the surface, especially the PBB + R treatment, led to significant improvement in TES by 43.0 % compared to CT. This study also indicated that the MES amounted to only 42.6–49.3 % of TES, and the remaining 50.7–57.4 % was accounted for by NMES, i.e., regulating ES, supporting ES, cultural ES, and EDS. Overall, the method proposed in this study can deliver improved theoretical and policy insights into ES estimation as well as the identification of a novel management practice for higher ES and sustainability under the maize – wheat cropping system in the IGP region of South Asia. This will sensitize policymakers and stakeholders to the importance of ES, the value accrued to society, and the need to maintain and enhance it. There is also a need for more studies at the local and regional levels to represent site-specific characteristics of ES and the development of methodology for the payment of TES. Efficient agricultural management practices are keys to realizing the benefits of ecosystem services and reducing dis services from agricultural activities. CA-based farming has higher economic values of ecosystem services, highlighting the long-term sustainability of these practices. By adopting sustainable practices and valuing ecosystem services, agriculture can contribute to a healthier planet and a more secure food future. The economic valuation of different ecosystem service showed that higher payment of ES could be obtained through sustainable farming based on CA over conventional method of farming. Based on the result of the study, a payment scheme for ES can be formulated by the policy makers.

CRediT authorship contribution statement

Nandita Mandal: Writing – original draft, Methodology, Formal analysis, Data curation. **Pragati Pramanik Maity:** Writing – review & editing, Writing – original draft, Supervision, Resources, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization. **T.K. Das:** Writing – review & editing, Writing – original draft, Resources, Investigation, Funding acquisition, Conceptualization. **K.K. Bandyopadhyay:** Writing – review & editing, Resources. **Sujan Adak:** Writing – review & editing, Formal analysis. **Abhradip Sarkar:** Writing – review & editing, Formal analysis, Data curation. **Ranjan Bhattacharyya:** Writing – review & editing. **Suman Sen:** Writing – review & editing, Data curation. **Subash N. Pillai:** Writing – review & editing, Resources. **Bidisha Chakrabarti:** Writing – review & editing, Resources.

Declaration of competing interest

The authors do not have any conflict of Interest.

Data availability

Data will be made available on request.

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