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A general framework for the quantification and valuation of ecosystem services of tree-based intercropping systems

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Abstract This study provides the first complete framework for the valuation of ecosystem services of agroforestry and uses a tree-based intercropping (TBI) system in southern Québec, Canada, as a case study. Ten ecosystem services were estimated, all of which were of interest and directly applicable to most agricultural systems worldwide: nutrient mineralization, water quality, soil quality, pollination, biological control, air quality, windbreak, timber provisioning, agriculture provisioning, and climate regulation. A mix of mathematical models for the quantification and economic valuation of various ecosystem services were used. The results revealed a total annual margin of \$2,645 ha^{-1} y^{-1} (averaged over 40 years). The economic value of combined non-market services was \$1,634 ha^{-1} y^{-1}, which was higher than the value of marketable products (i.e. timber and agricultural products). An analysis of the present value suggested that agricultural products ranked highest among the ecosystem services taken singularly, followed by water quality, air quality, climate regulation, and soil quality maintenance. Total economic value of all ecosystem services for the rotation period was \$54,782 ha^{-1}, only one third of which was contributed by agricultural products. Although the total value of the ecosystem services provided by TBI was high, farmers only benefited from agricultural and timber products. Thus, government incentives are needed to interest farmers in adopting practices that benefit society as a whole.

Keywords Economic valuation · Tree-based intercropping systems · Non-market benefits · Ecosystem services

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Introduction

Ecosystem goods and services (ES) are loosely defined as the benefits people obtain from ecosystems on earth (Millennium Ecosystem Assessment 2005). They include the provision of food and fiber, the regulation of water quality and pollination, supporting services such as soil formation, and a range of non-material benefits such as recreation and aesthetics. Agricultural systems are both producers and consumers of many of these services. While they produce much needed provisioning services for society, they also benefit from many of the regulating and supporting services produced within and outside the systems. Trade-offs are often observed since change in one service may lead to change in one or more of the others in the same or opposite direction. For example, the use of chemical fertilizers increases yield, often at the expense of pollution of soil and aquatic systems (Bennett et al. 2009).

Agricultural intensification to feed the ever-growing population of the world has raised environmental concerns such as soil erosion, water pollution, and degradation of biological diversity in agricultural landscapes. In view of these ecological problems related to conventional agriculture, a pressing question is how to simultaneously increase agricultural production while conserving a healthy and well-functioning life support system. Agroforestry has long been seen as an option to work at the interface of these global challenges (Nair and Garrity 2012). Studies have shown that this land use has the potential to maintain agricultural productivity, conserve biodiversity in agricultural landscapes as well as help mitigate climate change impacts (Udawatta and Jose 2012; Aertsens et al. 2013). Despite the demonstrated contribution of agroforestry in producing these ecological services, economic analyses on non-market services, as well as on the potential trade-offs between bundles of services, are little or non-existent. Some studies provide a general account of the role of agroforestry systems in providing ecosystem services (Jose 2009), while others provide frameworks for cost-benefit analysis of tropical agroforestry systems (Alavalapati and Mercer 2004). However, a comprehensive analytical framework for quantifying and valuing ES is missing in the context of temperate systems.

Although agroforestry is well established and even an old tradition practised elsewhere in the world with acknowledged positive effects (Rivest et al. 2013a), its practice in North America is only recent. In many parts of the continent, especially in Canada, agroforestry exists in the form of windbreaks, silvopasture, riparian buffers, forest farming, and alley cropping. Alley cropping, also known as tree-based intercropping (TBI), is the cultivation of agricultural products in between rows of trees. Such a system is under experimentation since 1987 with the establishment of a 30 ha plot in southern Ontario (Thevathasan and Gordon 2004). More recently, several experimental plots have been established in Québec (Rivest and Olivier 2007). Tree species in these plantations include, among others, hybrid poplar (*Populus spp*), black walnut (*Juglans nigra*), red oak (*Quercus rubra*), white ash (*Fraxinus Americana*), and Norway spruce (*Picea abies*) representing trees of fast-, medium-, and slow growth potentials. Field crops experimented include winter wheat, barley, corn and soybean, as well as fodder. Tree density varies depending on sites and age (e.g. 111 stems/ha in Guelph (Ontario) and 313 stems/ha before thinning in St- Rémi (Québec). Details of experimental set up, species composition, management regime and results from TBI systems in Canada can be found in Rivest et al. (2010), Thevathasan and Gordon (2004), Peichl et al. (2006) and Oelbermann et al. (2006).

The results obtained so far have shown that tree-based intercropping holds a great potential in providing a number of environmental services including reduction of nutrient leaching (Bergeron et al. 2011), enhancement of soil nutrient status (Rivest et al. 2009), increase of soil microbial community and tree growth (Rivest et al. 2010, 2013b) and sequestration of carbon (Evers et al. 2010). Tree-based intercropping is also likely to contribute many other important ES, but their quantification is difficult and, more importantly, they do not yet provide direct private benefits (i.e. benefits that can be measured through immediate market transactions) to the farmers. It is unlikely that farmers will deliberately adopt a new technology unless it is proven to be more profitable, or if they obtain compensation for services provided to the society. Simpson (1999) has shown that TBI systems may reduce farmers' private benefits to some extent in certain situations, while providing many public benefits. Although we can easily measure private benefits in terms of the market value of agricultural products, there is a lack of economic metrics to evaluate public benefits and, as a result, these benefits are referred to as 'externalities' and never accounted for in traditional cost-benefit analyses. This is also true for other natural systems, the reason being the lack of appropriate tools to measure the services and aggregate them in economic terms. We have only recently started to evaluate the monetary contribution of earth's biomes to human well-being (De Groot et al. 2012). However, the economic contribution of agroforestry systems is not known yet, despite the widespread recognition of agroforestry as a land use of high potential for provision of a multitude of products and services (Nair and Garrity 2012).

Therefore, the question is if we are to bring these 'externalities' into the evaluation model, how do we evaluate their contributions in economic terms for an agroforestry system like TBI? Also, what is the monetary value of the societal benefits of TBI as opposed to private benefits of monoculture in agriculture? These are important questions needing answers to help formulate policies that could guide the development of any mechanisms such as 'payments for ecosystem services' to offset the private costs incurred by the farmers while they supply societal benefits. This article is an attempt to answer such important questions still left unanswered. This article also provides a general framework that can be used to quantify and monetize ES in other agroforestry and social-ecological systems of interest.

The article is structured as follows: the following section provides the analytical framework, including the selection of ES, site characteristics, planting specifications and evaluation method. We then quantify and monetize annual margins of individual ecosystem services. We then summarize marginal values, evaluate economic value of ES and analyse trade-offs between various bundles of ES. Finally we extrapolate plot-scale findings in the Province of Québec, and discuss results and policy options.

Analytical framework

We evaluated ten ES for tree-based intercropping systems. The overall objectives included a marginal analysis of economic value of ES as well as the evaluation of the present value of future provision of services over a period of 40 years. Although rotation is determined by the objectives of the system, and can be as short as 15 years for some fast-growing trees and low-grade forest products (e.g. chips), we performed our analysis for 40 years to capture longer-term economic uncertainties. We made use of a 4-step analytical framework (Fig. 1). In the first step, we identified the full suite of ES which are meaningful in the context of the study. In doing so, we made an inventory of all possible ES from agroforestry; then, based on consultation with expert colleagues and literature reviews, we short-listed 10 services for analysis. In the second step, we quantified the service providing units and their relationships with the provision of services. In the third step, we attempted economic valuation of each of the ES. The final step involved extrapolation of results and examining trade-offs.

We used a mix of mathematical models for quantification of various ES and their economic valuation. In some instances, we used already existing models and equations, but in most instances we modified existing models or developed new ones to meet our needs. We used data published from experiments in various TBI sites in Québec and Ontario for most cases. In a few other cases we transferred data from study sites situated elsewhere.

The final list of ten ES included: nutrient mineralization (ES_1), water quality (ES_2), soil quality (ES_3), pollination (ES_4), biological control (ES_5), air quality (ES_6), windbreak (ES_7), timber provisioning (ES_8), agriculture provisioning (ES_9) and climate regulation (ES_{10}). We used the following sets of general equations for economic analysis:

$$TEV = \sum ES_n = \sum ES_{non-market} + \sum ES_{market}$$

where n = 1, 2, 3,… 10, the individual ecosystem service (ES), TEV = Total economic value, $\sum ES_{\text{non-market}} = \sum ES_{1\text{–}7,\,10}$ and $\sum ES_{\text{market}} = \sum ES_{8,\,9.}$

Below we provide methods of quantification of individual ES along with economic data and assumptions associated with the evaluation.

Assessment of annual margins

Nutrient mineralization

Nitrogen (N) release from poplar litter fall is reported to be 7 kg ha^{-1} y^{-1} (Thevathasan and Gordon 2004), which means annual fertilizer cost could be saved for

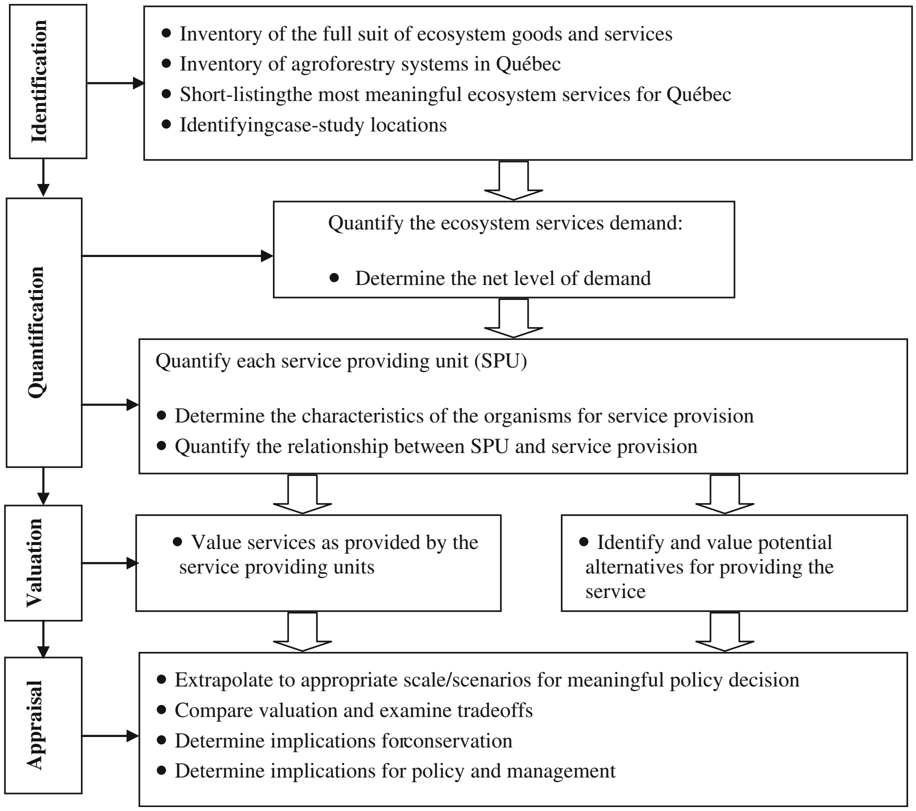

Fig. 1 Analytical framework for evaluation and valuation of ecosystem services of TBI systems

that amount. Based on the results from an experimental plot with a density of 111 trees ha^{-1} Zhang (1999) suggested a phosphorus (P) input by hybrid poplars of 4.99 kg ha^{-1} y^{-1} (in terms of through fall (1.06 kg ha^{-1} y^{-1}), litter fall (0.38 kg ha^{-1} y^{-1}) and net stem flow (3.56 kg ha^{-1} y^{-1}). Potassium (K) input was reported to be of 21.22 kg ha^{-1} y^{-1}; the equivalent amount of fertilizer for the corresponding K inputs, therefore, would be 11.42 kg ha^{-1} y^{-1}.

Rivest et al. (2009) observed in St-Rémi (stand density 313 stems/ha) and St-Édouard (stand density 419 stems/ha) that the above-ground biomass of hybrid poplar trees associated with various intercrops was 40 % higher on average than what was observed in controls without intercrop after 3–4 years of establishment. As mineralization of soil nutrients contributes to plant available nutrient, it is thus reasonable to assume that a certain percentage of tree yields are attributable to nutrient inputs and soil management of the system. If we conservatively assume that 10 % of mean annual increment in biomass is attributable to nutrient mineralization through the system, we are then able to estimate monetary contribution through the market price of that biomass. Field data from St-Édouard reveals a biomass increase of 20.3 kg/tree in 4 years (Rivest et al. 2009). Extrapolating this figure for all trees of 1 ha land area, we get annual biomass increment per hectare, which is equivalent to 1.62 m^3 ha^{-1} y^{-1}. Market prices of various nutrients used include: N, \$554 ton^{-1}; P, \$665 ton^{-1}; K, \$647 ton^{-1} (Toor et al. 2012; USDA 2013[1]).

Water quality

We evaluated water quality services in terms of cost of decontamination of nutrient loads as well as of the sediment dredging cost using the following equation:

[1] At the time of calculation the exchange rate between Canadian dollars and USD was very close to 1. For example the average exchange rates of 1USD in January 2013 were between 0.9919 and 1.0081 CAN$.

$$V_{water} = L_N \times C_{dN} + L_P \times C_{dP} + S \times C_{dred}$$

where $V_{water,}$ is the value of water quality regulation, L_N is the rate of N leaching reduced, C_{dN}, cost of N decontamination, L_P is the rate of P leaching, C_{dP} is the cost of P decontamination, S is the sedimentation rate and C_{dred} is the dredging cost.

Nitrogen leaching losses have been estimated to be approximately 9 kg ha^{-1} y^{-1} at the intercropping site whereas leaching losses in a monocropped field adjacent to the intercropped field were 20 kg N ha^{-1} y^{-1} (Thevathasan 1998). Therefore, intercropping appears to have reduced leaching losses by 11 kg N ha^{-1} y^{-1}. Further, MacDonald and Bennett (2009) estimated that the average P fertilizer application rate in nine watersheds of southern Québec exceeded crop requirements by 15–22 kg ha^{-1}. Since N leaching reduction was approximately 50 % in TBI, we hypothesize similar potential for P reduction. Consequently, 7.5 kg P ha^{-1} was assumed to be trapped in the system.

Costs of removing excess nutrients in waste treatment plants were reported to be \$8.50/kg for N and \$61.20/kg for P (Olewiler 2004), and erosion control and sediment retention by pasture lands, hedgerows and cultural woodlands (i.e. agricultural land) are worth an estimated \$5.60 ha^{1} y^{-1} (Wilson 2008a).

Soil quality

Following Sandhu et al. (2008) soil quality regulation was assessed in terms of soil formation. Sandhu et al. is thought to be the only study that modeled soil fertility based on direct measurement of the indicators (Crossman et al. 2013). Based on earthworms and other soil invertebrate data, the amount of soil formed was calculated, which was then multiplied by market price of soils. The equation can be expressed as:

$$\begin{aligned} V_{SoilF} &= (Q_{earth} + Q_{invert}) \times P_{soil} \\ &= (N_{earth} \times 0.0002 + Q_{invert}) \times P_{soil} \end{aligned}$$

where V_{SoilF} is the price of soil produced ha^{-1} y^{-1}, Q_{earth} is the amount of soil formed by earthworms, Q_{invert} is the amount of soil formed by invertebrates, P_{soil} is the market price of soil (\$ ton^{-1}), N_{earth} is the number of earthworms in the soil and 0.0002 is the weight of 1 earthworm (kg).

In this equation the weight of 1 earthworm equals 0.2 g and 1 ton of earthworm produces 1,000 kg soils ha^{-1} y^{-1} (Sandhu et al. 2008). Price and Gordon (1999) suggested that the number of earthworms equals 119–394 m^{-2} and biomass equals 245–557 g m^{-2} in poplar intercropping. If we assume that biomass of earthworms is 250 g m^{-2}, then there is 2.5 ton of earthworm biomass per hectare. If 1 ton earthworms produces 1,000 kg soils ha^{-1} y^{-1} (Sandhu et al. 2008), then total soils produced in poplar agroforestry is 2.5 ton ha^{-1} y^{-1}. Further, the contribution of soil invertebrates in soil formation is 1 ton ha^{-1} y^{-1} (Pimentel et al. 1995, 1997).

A market survey reveals a high variability in the price of top soil from as low as \$50 ton^{-1} to as high as \$300 ton^{-1}. We have used the lower bound market price of soil in our analysis.

Pollination

There are several methods of getting the value of pollination services. 'Replacement cost method' looks at how much the farmer spends to replace natural pollination with pollination by rental bees. Thus pollination service value could be obtained by multiplying area under crop production (excluding area under tree management in agroforestry) by industry-wide recommended honeybee stocking (e.g. 1 hive ha^{-1} for canola) and rental price of honey bee (Winfree et al. 2011). The second approach was recommended by Moradin and Winston (2006). This study found that pollinator abundance was greatest in the canola fields that had more uncultivated land within 750 m of field edges. A cost-benefit model in the study estimated that yield and profit could be maximized with the presence of a 30 % trees and shrubs cover in agricultural landscapes. The third approach, which we use in this study, is the 'production function approach'. Morse and Calderone (2000) used the following equation to estimate the value of honey bee in crop pollination:

$$V_{hb} = \sum (V \times D \times P_{hb})$$

where V_{hb} is the sum of the total annual value of insect pollinated crops that are pollinated by honey bees, V is the annual value of each crop, D is the dependency of each crop on insect pollinators, and P_{hb} is the estimate of the proportion of the effective insect crop pollinators that are honey bees.

We modified this function assuming that a single crop will be under evaluation and thereby avoiding 'summation'. We further excluded proportion of honeybee P_h from the equation since we are accounting contribution of all pollinating agents, as opposed to a single insect group. Additionally, we deducted variable costs from the revenue to attribute pollinators' contribution in the net profit. To avoid complexities in calculations we excluded timber management costs from the variable cost. In calculating yield per hectare we used exact land area under crop by deducting the area under tree management in the agroforestry plot. The final equation took the following form:

$$\mathrm{ESV_{pol}} = (\mathrm{Y.P} - \mathrm{VC}) \cdot \mathrm{D}$$

Here, $\mathrm{ESV_{pol}}$ is the Ecosystem Services Value of pollination, Y is the soybean yield = 1.47 ton $\mathrm{ha^{-1}\ y^{-1}}$, P is the soybean price = \$533.97 $\mathrm{ton^{-1}}$, VC is the variable cost = \$554 $\mathrm{ha^{-1}}$ (Toor 2010; Toor et al. 2012), and D is the pollinator dependence for soybean = 0.1 (Morse and Calderone 2000).

Biological control

An economic model based on the difference in the proportion of berries infested by berry-borer between exclosure and control plants estimated an average benefit of \$75 $\mathrm{ha^{-1}}$ with a range of \$44 to \$105 $\mathrm{ha^{-1}\ y^{-1}}$ (Kellermann 2007). Calculations of the benefits provided here were obtained by documenting pest infestation levels in the presence and absence of bird foraging (via exclosures) and translating higher saleable crop yields in the presence of birds into a dollar figure using crop market prices. We used the average value (i.e., \$75 $\mathrm{ha^{-1}}$) in our analysis.

Air quality

In 'contingent valuation approach' local residents are questioned on their willingness to pay (WTP) for a certain level of improved air quality enhanced by agroforestry. However most agricultural landscapes in Québec are located away from large urban settlements. A scarce population will also result in scarcity in air quality appreciation, therefore willingness to pay will not make sense. Hedonic pricing could be another option, but is faced with the same limitation as with contingent valuation. The remaining option was 'alternative cost of pollutant removal'. The role of trees in removing air pollutants such as NO_2, SO_2, dust and other particulate matter has been assessed by many researchers (Dwyer et al. 1992; Nowak et al. 2006; McPherson et al. 1999). Weathers et al. (2001) compared the concentration of chemical compounds (e.g. sulphur, nitrogen and calcium) between forest edges, forest interior and adjacent fields. Concentrations of the compounds were higher in forest edges, indicating that edges of forest or windbreak can be effective in filtering polluted air. Dawyer et al's study in urban forestry context in California found that 90,000 urban trees removed 154 tons of particulate matter annually. This corresponds to the removal of 1.67 kg pollutants by a single tree per year. Agroforestry landscapes, however, are not found in urban areas, and thus the same rate of pollutant removal is unlikely. Arbitrarily assuming the air quality maintenance service provided by each tree in an agroforestry plot to be a removal of 0.67 kg pollutants per tree and assuming per kilogram removal cost of \$6.29 (Wilson 2008a), a single tree provides a service worth \$4.20 per year. In a 110 trees $\mathrm{ha^{-1}}$ plot we obtain the annual air quality maintenance service provided by agroforestry by multiplying the dollar amount (i.e. \$4.20) with the total number of trees.

Windbreak

Tree belts established between crop fields around agricultural infrastructure, around livestock barns or close to residential infrastructures provide services through several mechanisms, including enhancing microclimate and conserving the natural environment. They also act as a barrier against pesticide drift (Ucar and Hall 2001), increase agricultural productivity in providing crops with shelters against wind storms and better snow management in the crop field (Jairell and Schmidt 1999), save energy cost when maintained around livestock and residential infrastructure (Wang 2006), and enhance overall animal wellbeing (Jairell and Schmidt 1999). The economic value of windbreaks can be evaluated using the following equations.

$$\mathrm{EV_{wb(c)}} = \mathrm{EV_p}$$

$$\mathrm{EV_{wb(L)}} = \mathrm{EV_{e+}EV_w}$$

where, $\mathrm{EV_{wb(c)}}$ is the value of ecosystem services provided by windbreak in the crop fields, $\mathrm{EV_e}$ is the value of energy saved, $\mathrm{EV_p}$ is the value of overall increased productivity in agriculture due to reduction of wind erosion and snow management, EV_w is the value of overall animal wellbeing, $EV_{wb(L)}$ is the value of windbreak around livestock facility.

In our analysis we evaluated windbreak services in terms of overall increased productivity. In an earlier study Kort (1988) reported about 3.5 % increase in spring wheat due to the presence of windbreak. Later on, Brandle et al. (2004, 2009) showed that the overall increased productivity in agriculture due to reduction of wind erosion and snow management is 15–20 %. However, trees in the intercropping systems are widely spaced on the lines planted, but many more of them are installed across a given field. Therefore we do not exactly know how they contribute to wind control with respect to windbreaks. We conservatively assumed a 5 % increase in yield of 1.47 ton ha^{-1} (i.e. 73.5 kg ha^{-1}) attributable to windbreak.

Provisioning services

Valuation of provisioning services is relatively straightforward and can be accomplished in terms of provision of agricultural, timber and non-wood tree outputs. In this study, however, non-wood tree products such as medicines and fruits, firewood and intermediate thinning and pruning products were excluded. Following Toor et al. (2012) we used data on crop yield (soybean) of 1.47 ton ha^{-1} y^{-1}, timber yield (hybrid poplar) of 3.5 m^3 ha^{-1} y^{-1}, with crop market price of \$533.97 ton^{-1} and timber market price of \$40 m^{-3}.

Climate regulation

Net carbon sequestration from an agroforestry plot can be estimated as the sum of above ground C sequestration plus below ground C sequestration less carbon liberation into atmosphere through various processes. For operational purpose the equation for C sequestration accounting can be written as:

$$NCS = (B_t + B_r + B_l + CR + SOC) - (C_r + C_l) + C_{N2O}$$

where, NCS is the net carbon sequestered, B_t, and B_r is the carbon stored in tree trunk biomass (including branches and leaves) and roots respectively, B_l is the carbon stored in litter fall, CR is the carbon stored in crop residues, SOC is the carbon pool in soil, C_r, is the carbon returned back through soil respiration, C_l is the carbon lost through leaching into soil profiles, C_{N2O} is the CO_2 equivalent avoided emission of N_2O.

Table 1 C/CO_2 fluxes in tree-based intercropping systems

C/CO_2 fluxes	Amount ($Mg\ C\ ha^{-1}\ y^{-1}$)	Amount ($Mg\ CO_2e\ ha^{-1}\ y^{-1}$)
Total sequestration	6.86	25.2
Total release	4.6	16.88
Net sequestration	2.26	8.3

The above equation reveals total carbon sequestration potential of TBI to be of 6.86 Mg C ha^{-1} y^{-1}. Data indicate an above ground carbon sequestration of 4.16 Mg C ha^{-1} y^{-1}, while below ground estimate is 2.7 Mg C ha^{-1} y^{-1}, just over a quarter of above ground sequestration rates. Total carbon lost through leaching and soil respiration is higher than total below ground sequestration. Out of the total carbon sequestered 4.6 Mg C ha^{-1} y^{-1} will go back to the atmosphere through these processes. Hence net carbon sequestration potential is 2.26 Mg C ha^{-1} y^{-1} (Table 1). This amount of C represents immobilization of 8.3 Mg CO_2 (1 ton of carbon equals $44/12 = 3.67$ tons of carbon dioxide) from one hectare TBI plot in a year. We applied a social cost of carbon (SCC) value of \$43 (Yohe et al. 2007) in this analysis. The SCC, also referred to as Damage Cost Avoided, represents the marginal cost of emitting an additional unit of CO_2 into the atmosphere, i.e. the estimate of monetary value of damage resulting from CO_2 emissions.

A summary of various indicators, service providing units and marginal economic values described above can be found in Table 2.

Aggregation and extrapolation

In the above sections we have shown how to quantify and monetize ES in a TBI context. In this section we describe the net present value (NPV) of each of the services for 40 years and aggregate them in different bundles of services. While marginal benefit shows what the annual economic value of the services per unit area is, a NPV provides an understanding of how the benefit is observed over a longer time-frame, 40 years in this case. We did this by discounting the

Table 2 Indicators and economic values of ecosystem services of tree-based intercropping system

TBI ecosystem services	Indicators	Indicator quantity	Economic value ($ ha^{-1} y^{-1})	References
Nutrient mineralization	N input	7 kg ha^{-1} y^{-1}	3.8	Thevathasan and Gordon (2004); Zhang (1999); Rivest et al. (2009); Toor et al. (2012); USDA[a]
	P input	11.42 kg ha^{-1} y^{-1}	7.5	
	K input	21.22 kg ha^{-1} y^{-1}	13.5	
	Change in yield (timber)	0.162 m^3 ha^{-1} y^{-1}	6.4	
Water quality	N decontamination	11 kg ha^{-1} y^{-1}	93.5	Olewiler (2004)
	P decontamination	7.5 kg ha^{-1} y^{-1}	459	Olewiler (2004)
	Sediment dredging	–	5.6	Wilson (2008a)
Soil quality	Earthworms	2.5 ton ha^{-1} y^{-1}	125	Sandhu et al. (2008); Price (1999)
	Invertebrates	1 ton ha^{-1} y^{-1}	50	Pimentel et al. (1995, 1997)
Pollination	Yield changes (crop)	1.47 ton ha^{-1} y^{-1}	24.1	Morse and Calderone (2000); Toor et al. (2012)
Biological control	Pest infestation levels	–	75	Kellermann (2007)
Air quality	Pollutant removal	1.67 kg/tree	462	Wilson (2008a)
Windbreak	Productivity change	1.47 ton ha^{-1}	39.2	Brandle et al. (2004, 2009)
Timber provisioning	Annual yield	3.5 m^3 ha^{-1} y^{-1}	140	Toor et al. (2012)
Agriculture provisioning	Annual yield	1.47 ton ha^{-1} y^{-1}	784.9	Toor et al. (2012)
Climate regulation	Carbon sequestration	8.3 Mg CO_2e ha^{-1} y^{-1}	356.9	Unpublished data

[a] http://www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx#26727

future values into present values with a discount rate of 4 %.

The total annual margin of TBI ecosystem services was estimated to be $2,645 ha^{-1} y^{-1}. The economic value of combined non-market services was $1,634 ha^{-1} y^{-1}, which was higher than the value of marketable products (i.e. timber and agricultural products). The economic return from agriculture in monoculture was $1,110 ha^{-1} y^{-1}, whereas the return from agriculture in TBI was $785 ha^{-1} y^{-1}. Table 3 presents a breakdown of the marginal value of different bundles of ES stemming from TBI.

An analysis of the present value of future benefits of ES for the rotation of 40 years was also carried out. Provision of agricultural products ranked highest ($16,287 ha^{-1}) among the ES, followed by water quality ($11,581 ha^{-1}), air quality ($9,510 ha^{-1}), carbon sequestration ($7,346 ha^{-1}), and soil quality ($3,631 ha^{-1}) (Table 3). Total economic value of all the ES was $54 782 ha^{-1}, only a third of which was contributed by agricultural products. Total non-market benefits were twice as high as the provisioning services combined (i.e. timber and agriculture) (Table 4).

There is no precise estimate of how much of the available farms could be converted into agroforestry in Québec. Based on present production type (see below), Oelbermann et al. (2006) stated that 40 % of Canada's approximate 7 M ha of marginal lands are eligible to be converted into agroforestry, whereas spatial analysis done by Hernandez et al. (2008) showed that a 34 % increase in wooded area in the L'Ormière River watershed in Québec is possible through agroforestry practices. If we conservatively assume that 20 % of Québec's 1.93 M ha croplands can be converted to TBI, then the potential benefits of TBI ecosystem services are equivalent to about $5 billion per year. This figure excludes summer fallow land (4,288 ha), tame or seed pasture (147,387 ha), natural land for pastures (158,602 ha) and other land area including Christmas tree areas, woodlands and wetlands (> 1.2 M ha) (Statistics Canada 2006).

Discussion

This study provides the first estimate of economic values of ES generated by TBI systems. The values

Table 3 Breakdown of marginal and net present values of TBI ecosystem services

Ecosystem services	Marginal values ($ ha^{-1} y^{-1})	NPVs ($ ha^{-1})
Nutrient mineralization	31	652
Water quality	558	11,581
Soil quality	175	3,631
Pollination	24	500
Biological control	75	1,556
Air quality regulation	462	9,510
Windbreak	39	813
Timber provisioning	140	2,905
Agriculture provisioning	785	16,287
Climate regulation	356	7,346

Table 4 Ecosystem services in various bundles in tree-based intercropping systems

Bundles	Marginal values ($ ha^{-1} y^{-1})	NPVs ($ ha^{-1})
Agriculture in monoculture	1,110	23,046
Agriculture in TBI	784	16,287
TBI provisioning	924	19,192
TBI non-market	1,634	35,590
Total economic Value (TEV)	2,645	54,782

ranged from $24 ha^{-1} y^{-1} for pollination to $785 ha^{-1} y^{-1} for agricultural products. Water quality regulation ranked highest among the non-market services, followed by air quality regulation and carbon sequestration. Although conventional agriculture provides more private benefits than TBI, the value of ES of TBI to society is much higher compared to this private value. Such information concerning economic returns of TBI as a long-term investment is very important when establishing policies that would benefit the society as a whole.

The total potential value of TBI ecosystem services is estimated to be 5 billion dollars a year in Québec. Ecosystem services of TBI systems are not directly comparable to other systems. However, results of similar economic analysis in forested landscapes in Canada reveal values such as $2.6 billion per year for Ontario's green belt (Wilson 2008a), $0.9 billion for Lake Simcoe basin watershed (Wilson 2008b), and up to $130 million for the Pimachiowin Aki conservation area in Ontario (Voora and Barg 2008). A recent study estimated the value of natural capital, which includes ecosystem goods and services of forests, rivers, swamps and other natural areas, of a 1.7 million hectare land area surrounding Montréal (Québec), to be $3 billion a year (Dupras et al. 2013).

Although agroforestry in different forms and models exists in different parts of the world, the history of such farming systems is relatively new in Canada. However, many farmers in Canada are adopting agroforestry for farm and societal benefits. The 2006 census data reveals that in the province of Québec alone 5,994 farms out of 30,675 reported to have windbreaks, compared to 1,845 in 2001, an increase of more than 4,000 (Statistics Canada 2001, 2006). Such a trend in the adoption of trees in agricultural landscapes suggests that farmers could positively respond to TBI systems if they found them to be profitable. However, since the private benefits from TBI systems are less than the societal benefits in terms of provision of ES, government programs to subsidize farmers would be necessary to entice them to adopt TBI systems rapidly. The question, however, is determining what such programs could be? Payment for ecosystem services is regarded as an effective mechanism for managing sustainable provision of ecosystem services from landscapes and watersheds (Schomers and Matzdorf 2013; Ingram et al. 2014). Although most successful payment programs have been implemented in developing countries, there is evidence that such mechanisms can equally work in industrialized nations (Schomers and Matzdorf 2013). In the current context of agro-environment programs applicable in Québec, agroforestry practices are recognized and supported as are other agricultural beneficial management practices, essentially for specific ecological functions such as stabilizing river banks, reducing erosion and improving habitats for biodiversity. However, agroforestry systems differ from the majority of agricultural beneficial management practices in their ability to generate income through the production of various products and services possessing tangible economic value. For this reason, adopting programs focusing on both the private profitability of agroforestry practices and their public benefits is a fundamental issue. Addressing agroforestry through a multifunctional perspective would highlight its potential for regional socio-economic development through many issues such as economic diversification, development of new markets, job creation and retention of the rural workforce.

From a microeconomic point of view, the implementation of agroforestry practices by farmers is most often associated with a loss of income due to a reduction of the insurable crop acreage. The establishment of multifunctional and productive agroforestry systems changes the economic equation with the introduction of other income possibilities that could be an incentive for the adoption of agroforestry systems by farmers. Besides the one-time grant to implement a particular beneficial management practice, there are a variety of payments that could possibly be attached to various ecological functions supported by agroforestry, such as protection of water quality, habitat maintenance or carbon sequestration. Moreover, among all ES attributed to agroforestry, some may be conducive to a market approach. For example, a stock trading scheme for greenhouse gas is already operating in North America. The sale of carbon credits from agroforestry could be profitable for the producer. Other markets could be developed in the medium or long term, for example trading the emission of pollutant loads issued over water rights or related to biodiversity issues. This should lead to the development of political and financial support programs focusing on the production of ES, given their importance in an economic and environmental point of view.

Certain limitations of this study must be identified. First, there are more than one valuation approaches possible for every ecosystem service, each of which would give a different value. Value transfer is often assumed to be an appropriate method when there is insufficient time, funding and other resources to generate primary data. Therefore, the challenge is to select the most appropriate approach and model given the context of the study and the availability of required data. There are also certain limitations associated with the biophysical and economic estimates used in our study; for example, cutback in energy usage as a potential source of emission reduction was not accounted for. We were handicapped by a lack of sufficient quantitative data in the existing literature due to this relatively new practice in North America. Certain relevant studies, such as those conducted on tropical agroforestry systems, are of limited use to this study because of a completely different environmental setting. We were also disadvantaged by the lack of experimental data on various biophysical aspects. For example, data on air pollutant removal by trees in agricultural landscapes is non-existent, and as a result we arbitrarily assumed the potential of agroforestry trees in capturing pollutants based on existing results on urban forestry. These assumptions also bypassed the fact that the capacity of agricultural systems to remove pollutants depends on local biophysical and environmental settings (such as presence of forestry systems in adjacent areas, density of suspended particles in the air and so on). This, in turn, points to possible policy issues since subsidies would probably have to be determined with respect to the local or regional availability of other systems to provide the same services. Indeed, a farm converting to TBI in an otherwise intensively managed landscape would provide more services to the society than one located in a mostly forested landscape.

In the discourse on global climate change the economic values of the services presented here would be very different. Climate change is expected to have significant impacts on the yield and productivity of the agricultural sector in Québec and multifunctional TBI systems are seen as a potential adaptation option in this context (Domenicano 2013). The question then is how will the value of ES evolve with changing agricultural productivity in the future? We know from microeconomic theory that the price of a commodity increases as it becomes scarcer and demand is increased. Since the society's demand and willingness to pay for ES will increase, so will their value and price.

Finally, it should also be pointed out that an ecosystem develops over many years of interaction among its various components. Therefore, it may take years to start realizing benefits after establishing an agroforestry system. Besides, in this study we assumed a uniform distribution of the provision of the ES throughout the rotation period, which is certainly not the case in fact. Although certain components such as leaf litter distribution were found to be evenly distributed across crop alley in a 14 m tall poplar system (Thevathasan and Gordon 2004), there are ES indicators which can occur unevenly (see Bambrick et al. 2010). However, addressing such a complex issue was not within this study's scope. A large list of other services (such as those related to option value or bequest value) was not included in this study for simplicity's sake and general means of application. As a result, the value of ES of agroforestry in our study is probably an underestimate of the real monetary contribution of the system to farmers and the society.

Conclusion

This study enhances our understanding of the true value of market and non-market benefits of tree-based intercropping systems in temperate environments. It also provides a framework to quantify and monetize ecosystem services in other agroforestry systems. Despite inherent uncertainties in quantification and valuation of ecosystem services, which are non-market in nature, this study provides a reasonable estimate of the economic contribution of tree-based intercropping systems to society's welfare. The demonstrated benefits are substantial. However, in Québec context the management of TBI systems still needs to be optimized in order to make it more profitable for farmers than is conventional agriculture, as already observed in Europe. The benefits of their ecosystem services are realized at the cost of farmers' private benefits due to reduced provisioning services and the expected cost of adoption and maintenance of this new technology over a longer time frame. While it is impractical to suggest that all agricultural lands should be converted to agroforestry, a land inventory can determine the areas suitable for TBI based on environmental and technical feasibility and the willingness of the farmers in participating. Therefore, the adoption and expansion of TBI systems in Québec as well as in other parts of Canada is certainly worthy of discussion in policy forums.

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References

Aertsens J, Nocker LD, Gobin A (2013) Valuing the carbon sequestration potential for European agriculture. Land Use Policy 31:584–594

Alavalapati JRP, Mercer DE (eds) (2004) Valuing agroforestry systems. Advances in agroforestry, vol. 2, Kluwer Academic Publishers, Dordrecht

Bambrick AD, Whalen JK, Bradley RL, Cogliastro A, Gordon AM, Olivier A, Thevathasan NV (2010) Spatial heterogeneity of soil organic carbon in tree-based intercropping systems in Québec and Ontario, Canada. Agrofor Syst 79(3):343–353

Bennett EM, Peterson GD, Gordon L (2009) Understanding relationships among multiple ecosystem services. Ecol Lett 12:1–11

Bergeron M, Lacombe S, Bradley RL, Whalen J, Cogliastro A, Jutras M, Arp P (2011) Reduced soil nutrient leaching following the establishment of tree-based intercropping systems in eastern Canada. Agrofor Syst 83:321–330

Brandle JR, Hodges L, Zhou X (2004) Windbreaks in sustainable agriculture. Agrofor Syst 61:65–78

Brandle JR, Hodges L, Tyndall JC, Sudmeyer RA (2009) Chapter 5: Windbreak practices. In: Garrett HE (ed) North American agroforestry: an integrated science and practice, 2nd edn. American Society of Agronomy, Madison, pp 75–104

Crossman ND, Burkhard B, Nedkov S, Willemen L, Petz K, Palomo I, Drakou EG, Martin-Lopez B, McPhearson T, Boyanova K, Alkemade R, Egoh B, Dunbar MB, Maes J (2013) A blueprint for mapping and modelling ecosystem services. Ecosyst Serv 4:4–14

De Groot R, Brander L, van der Ploeg S, Costanza R, Bernard F, Braat L, Christie M, Crossman N, Ghermandi N, Hein L, Hussain S, Kumar P, McVittie A, Portela R, Rodriguez LC, ten Brink P, van Beukering P (2012) Global estimates of the value of ecosystems and their services in monetary units. Ecosystem Services 1(1):50–61

Domenicano S (2013) Using modeling to predict future scenarios: Will climate change drive agroforestry systems in temperate North America towards increased competition or complementarity? Paper presented at the 13th North American Agroforestry Conference., Charlottetown, Prince Edward Island, 19–21 June, 2013

Dupras J, Revéret JP, Michaud C (2013) The value of the ecosystem services in greater Montreal (Québec). In: Proceedings of the international conference of the european society for ecological economics

Dwyer JF, McPherson EG, Schroeder HW, Rowntree RA (1992) Assessing the benefits and costs or the urban forest. J Arboric 18(5):227–234

Evers AK, Bambrick A, Lacombe S, Dougherty MC, Peichl M, Gordon AM, Thevathasan NV, Whalen J, Bradley RL (2010) Potential greenhouse gas mitigation through temperate tree-based intercropping systems. Open Agric J 4:49–57

Hernandez M, Charland P, Nolet J, Arès M (2008) Carbon sequestration potential of agroforestry practices in the L'Ormière River watershed in Québec. Agriculture and Agri-Food Canada, Québec. ISBN 978-0-662-47230-8

Ingram JC, Wilkie D, Clements T, McNab RB, Nelson F, Baur EH, Sachedina HT, Peterson DD, Foley CAH (2014) Evidence of Payments for Ecosystem Services as a mechanism for supporting biodiversity conservation and rural livelihoods. Ecosyst Serv. doi:10.1016/j.ecoser.2013.12.003

Jairell RL, Schmidt RA (1999) Snow management and windbreaks. In: Proceedings of the range beef cow symposium XVI, Greeley, 14–16 Dec 1999

Jose S (2009) Agroforestry for ecosystem services and environmental benefits: an overview. Agrofor Syst 76:1–10

Kellermann JL (2007) Ecological and economic services provided by birds on Jamaican blue mountain coffee farms. MSc thesis, Humboldt State University, Arcata

Kort J (1988) Benefits of windbreaks to field and forage. Agric Ecosyst Environ 22(23):165–190

MacDonald GK, Bennett EM (2009) Phosphorus accumulation in the Saint Lawrence River watershed: a century-long perspective. Ecosystems 12:621–635

McPherson EG, Simpson JR, Peper PJ, Xiao Q (1999) Benefit-cost analysis of Modesto's municipal urban forest. J Arboric 25:235–248

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: synthesis. Island Press, Washington DC

Morandin LA, Winston ML (2006) Pollinators provide economic incentive to preserve natural land in agroecosystems. Agric Ecosyst Environ 116(3–4):289–292

Morse R, Calderone NW (2000) The value of honey bees as pollinators of U.S. Crops in 2000. Bee Cult 128:1–15

Nair PK, Garrity D (eds) (2012) Agroforestry—the future of global land use. Advances in Agroforestry 9, Springer, New York. doi 10.1007/978-94-007-4676-3_16

Nowak DJ, Crane DE, Stevens JC (2006) Air pollution removal by urban trees and shrubs in the United States. Urban For Urban Green 4:115–123

Oelbermann M, Voroney RP, Thevathasan NV, Gordon AM, Kass DCL, Schlönvoigt AM (2006) Soil carbon dynamics and residue stabilization in a Costa Rican and southern Canadian alley cropping system. Agrofor Syst 68(1):27–36

Olewiler N (2004) The value of natural capital in settled areas of Canada. Ducks Unlimited Canada and the Nature Conservancy of Canada, Regina

Peichl M, Thevathasan NV, Gordon AM (2006) Carbon sequestration potentials in temperate tree-based intercropping systems, southern Ontario, Canada. Agrofor Syst 66:243–257

Pimentel D, Harvey C, Resosudarmo P, Sinclair K, Kurz D, McNair M, Crist S, Shpritz L, Fitton L, Saffouri R, Blair R (1995) Environmental and economic costs of soil erosion and conservation benefits. Science 267:1117–1123

Pimentel D, Wilson C, McCullum C, Huang R, Dwen P, Flack J, Tran O, Saltman T, Cliff B (1997) Economic and environmental benefits of biodiversity. Bioscience 47:747–757

Price GW, Gordon AM (1999) Spatial and temporal distribution of earthworms in a temperate intercropping system in southern Ontario, Canada. Agrofor Syst 44:141–149

Rivest D, Olivier A (2007) Cultures intercalaires avec arbres feuillus : quel potentiel pour le Québec? For Chron 83(4):526–538

Rivest D, Cogliastro A, Olivier A (2009) Tree-based intercropping systems increase growth and nutrient status of hybrid poplar: a case study from two Northeastern American experiments. J Environ Manag 91:432–440

Rivest D, Cogliastro A, Bradley RL, Olivier A (2010) Intercropping hybrid poplar with soybean increases soil microbial biomass, mineral N supply and tree growth. Agrofor Syst 80:33–40

Rivest D, Paquette A, Moreno G, Messier C (2013a) A meta-analysis reveals mostly neutral influence of scattered trees on pasture yield along with some contrasted effects depending on functional groups and rainfall conditions. Agric Ecosyst Environ 165:74–79

Rivest D, Lorente M, Olivier A, Messier C (2013b) Soil biochemical properties and microbial resilience in agroforestry systems: effects on wheat growth under controlled drought and flooding conditions. Sci Total Environ 465–464:51–60

Sandhu HS, Wratten SD, Cullen R, Case B (2008) The future of farming: the value of ecosystem services in conventional and organic arable land. An experimental approach. Ecol Econ 64:835–848

Schomers S, Matzdorf B (2013) Payments for ecosystem services: a review and comparison of developing and industrialized countries. Ecosyst Serv 6:16–30

Simpson JA (1999) Effects of shade on maize and soybean productivity in a tree-based intercropping system. PhD Thesis, University of Guelph,Guelph

Statistics Canada (2001) Census of agriculture 2001. http://www.statcan.gc.ca/. Accessed 12 Feb 2013

Statistics Canada (2006) Census of agriculture 2006. http://www.statcan.gc.ca/. Accessed 12 Feb 2013

Thevathasan NV (1998) Nitrogen dynamics and other interactions in a tree-cereal intercropping systems in southern Ontario. PhD Thesis. University of Guelph, Guelph

Thevathasan NV, Gordon AM (2004) Ecology of tree intercropping systems in the North temperate region: experiences from southern Ontario, Canada. Agrofor Syst 61–62:257–268

Toor IA (2010) Economic analysis of tree-based intercropping in southern Ontario, Canada. M.Sc. Thesis, McGill University, Montreal

Toor IA, Smith EG, Whalen JK, Naseem A (2012) Tree-based intercropping in Southern Ontario, Canada. Can JAgric Econ 60(2):141–154

Ucar T, Hall FR (2001) Windbreaks as a pesticide drift mitigation strategy: a review. Pest Manag Sci 57(8):663–675

Udawatta RP, Jose S (2012) Agroforestry strategies to sequester carbon in temperate North America. Agrofor Syst 86: 225–242

USDA (2013) Fertilizer use and prices. Economic Research Service, United States Department of Agriculture, http://www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx#26727. Accessed 5 Feb 2013

Voora V, Barg S (2008) Pimachiowin Aki world heritage project area ecosystem services valuation assessment, http://www.iisd.org/pdf/2008/ecosystem_valuation.pdf. Accessed 21 Sept 2012

Wang F (2006) Modelling sheltering effects of trees on reducing space heating in office buildings in a windy city. Energy Buildings 38(12):1443–1454

Weathers KC, Cadenasso ML, Pickett STA (2001) Forest edges as nutrient and pollutant concentrators: potential synergisms between fragmentation, forest canopies and the atmosphere. Conserv Biol 15:1506–1514

Wilson SJ (2008a) Ontario's wealth, Canada's future: Appreciating the value of the greenbelt's eco-services. David Suzuki Foundation, Vancouver. ISBN 978-1-897375-17-4

Wilson SJ (2008b) Lake Simcoe Basin's Natural Capital: the value of the watershed's ecosystem services. David Suzuki Foundation, http://www.davidsuzuki.org/publications/downloads/2011/Lake-Simcoe-GreenbeltNaturalCapitalJune%20202.pdf. Accessed 21 Sept 2012

Winfree R, Gross BJ, Kremen C (2011) Valuing pollination services to agriculture. Ecol Econ 71:80–88

Yohe GW, Lasco RD, Ahmad QK, Arnell NW, Cohen SJ, Hope C, Janetos AC, Perez RT (2007) Perspectives on climate change and sustainability. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE (eds) Climate change 2007: impacts, adaptation and vulnerability: contribution of working group ii to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, pp 811–841

Zhang P (1999) The impact of nutrient inputs from stemflow, throughfall, and litterfall in a tree-based temperate intercropping system, southern Ontario, Can- ada. MSc Thesis, Dept. of Environmental Biology, University of Guelph, Guelph