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Economic valuation of suspended sediment and phosphorus filtration services by four different wetland types: A preliminary assessment for southern Ontario, Canada

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Abstract

Wetlands are known for their water filtration (or purification) function. Although different wetland types differ in their filtration capacity, they are usually aggregated together in economic valuation studies. Here, we explicitly separate the valuation of the suspended sediment and phosphorus (P) filtration services of the four major wetland types—bogs, fens, marshes and swamps—found in southern Ontario, Canada. The areal extents of the four wetland types are derived from the Canadian Wetland Inventory (CWI) progress map, while the sediment accretion rate is used as the key variable regulating the suspended sediment and P filtration functions. Based on available literature data, we assess the relationship of the sediment accretion rate to wetland size. Because only weak positive correlations are found, we assign a mean (average) sediment accretion rate to each wetland type. The sediment accretion rates are combined with mean soil P concentrations to estimate Pretention rates by the wetlands. The replacement cost method is then applied to valuate the sediment and P filtration services. The unit values for both sediment and P retention decrease in the order: marshes > bogs ≈ swamps > fens. The total value of sediment plus phosphorus removal by all wetlands in southern Ontario amounts to \$4.2 ± 2.9 billion per year, of which about 80% is accounted for by swamps. We further assess the costs of different options to offset the additional P loading generated in a hypothetical scenario whereby all wetlands are converted to agriculture. The results demonstrate that replacing the P filtration function of existing wetlands with conventional land management and water treatment solutions is not cost-effective, hence reinforcing the importance of protecting existing wetlands.

KEYWORDS
economic valuation, filtration service, phosphorus, sediment accretion, southern Ontario, wetland types

1 | INTRODUCTION

Wetlands, which are among the most productive terrestrial ecosystems, provide huge economic benefits through a variety of functions (Gallant et al., 2020). Among these, their hydrological (e.g., flood control), biogeochemical (e.g., nutrient retention) and ecological (e.g., nursery plants) functions deliver socio-economic benefits known as ecosystem services (Aziz & Van Cappellen, 2020). Differences in hydrological and geomorphological characteristics distinguish the various wetland types (Warner & Rubec, 1997; Table 1) that, in turn, results in variable provisioning of ecosystem services (Turner et al., 2000).

Wetlands have long been recognized for their key function of filtering pollutants out from water (Gopal & Ghosh, 2008). Increased sediment loads and nutrient enrichment are major threats to the quality of aquatic ecosystems (Dordio et al., 2008; Fennessy et al., 2004). Therefore, the role of wetlands in improving water quality is a primary argument for their preservation and restoration across the world (Bring et al., 2020; Dordio et al., 2008). Freshwater wetlands trap sediment and sequester nutrients (Craft & Casey, 2000) and filter water through physical (sedimentation), chemical (adsorption, precipitation, chelation) and biological (plant uptake) processes (Fennessy et al., 2004; Kadlec & Wallace, 2009; Kidd et al., 2015; Reddy et al., 1999; Settlemyre & Gardner, 1977).

Sediment deposition depends on wetland type with some types more efficiently retaining sediment than others (Bruland, 2008; Loaiza & Findlay, 2008). The sediment filtering effectiveness also depends on watershed size, land use and the wetland's connectivity to the stream and groundwater network (Craft & Casey, 2000). Sediment accumulation in wetlands is heavily affected by human activity in the watershed. For example, in the Murray–Darling Basin in Australia sedimentation rates doubled after European settlement and are now 80 times higher than the mean rate in the Late Holocene (Gell et al., 2009).

Sediment accretion is the net balance between sediment deposition and resuspension (Neubauer et al., 2002), and an important indicator of the functioning of restored wetlands (Takekawa et al., 2010). It is influenced by, among other things, the amount of suspended material delivered to the wetland, the composition and distribution of vegetation, flooding and waterlogging patterns, depth and bottom morphology and biomass production (Cahoon & Turner, 1989; Goodman et al., 2007; Jarvis, 2010). Sediment accretion rates for wetlands, however, are often difficult to estimate and data are relatively sparse (Loaiza & Findlay, 2008).

Through sediment retention, wetlands can be helpful in mitigating excess nutrients and pollutants (Mitsch & Gossilink, 2000). Hence, a wetland's sediment accretion rate is a critical parameter regulating water quality improvement (Bhomia et al., 2015; Gustavson & Kennedy, 2010). Wetlands remove phosphorus (P) from the water through physical and biological processes (Reddy et al., 1999). Phosphorus may accumulate in sediments by settling of allochthonous particulate P and autochthonous biomass P, the precipitation of aqueous

TABLE 1 Major wetland types and their characteristics (from National Wetlands Working Group, 1997; Smith et al., 2007; Zoltai & Vitt, 1995)

	Wetland type			
Attribute	Marshes	Swamps	Bogs	Fens
Definition	Shallow water areas that are mostly grasslands, can be freshwater or saltwater, amount of water in a marsh can change seasonally or with tide	Slow moving streams, rivers or isolated low areas with more open and deeper water than marshes	Peat lands raised or level with surrounding terrain; unaffected by runoff or groundwater from surrounding; receive water from precipitation; water table is at or slightly below surface	Peat land with fluctuating water table at surface, water channels enter in and water seeps through peat
Soil	Low mineral soil but substantial content of organic matter and nutrient rich	Poorly drained and water logged soil but nutrient rich	Low nutrient soils, peat is waterlogged, poorly oxygenated or devoid of oxygen	Solis have higher concentration of minerals than bogs and are nutrient rich
Moisture regime	Hydric to very hydric	Hygric to hydric	Subhygric to hygric	Hygric to hydric
pH	5.2–6.4	5.9–6.1	3.5–3.6	4–6.2
Vegetation	Freshwater marshes contain soft stemmed and non-woody plants, for example, grasses and shrubs, saltwater marshes have grasses, reeds, and rushes	Have woody shrubs and trees rather than grasses and herbaceous vegetation	May be treed or treeless, usually covered with Sphagnum spp. and shrubs which can survive in humid and nutrient poor conditions	Wetter fens are dominated by graminoid, bryophytes, sedge, rushes and moss vegetation, drier fens are dominated by trees as black spruce and shrubs
Morphology	Channel, coastal, shore, estuarine, kettle, stream, floodplain, and so on	Basin, flat, spring, stream, shore, peat margin, and so on	Basin, blanket, domed, flat, floating, mound, and so on	Basin, channel, floating, feather, spring, stream, and so on

P with metal cations, plus P sorption to mineral and organic substrates (Mitsch & Gossilink, 2000). Wetlands generally trap phosphorus although sometimes they may release aqueous P under anoxic conditions (Johnes et al., 2020). While uptake by vegetation can temporally remove P from water, P in accreted sediments represents the long-term sink in wetlands (Mitsch & Gossilink, 2000). Therefore, the sediment accretion rate is the key parameter used to estimate P retention in wetlands (Griffiths & Mitsch, 2020).

Wetlands are complex and diverse ecosystems, and therefore valuation of their ecosystem services is challenging. Economic valuation of some services relies on perceived benefits and people's preferences, which can vary significantly. Thus, there is no standard valuation framework as yet to value ecosystem services generated from different wetland types (Lambert, 2003). Nonetheless, studies imply that, in many areas of the world, ecosystem services have been declining due to draining of wetlands (Zedler, 2003). Since 1900, 50% of wetland areas have been lost worldwide. In southern Ontario, Canada, about 68% of wetlands have been converted to other uses since 1980 (Ducks Unlimited Canada, 2010). These huge losses in part reflect a lack of recognition of the economic value of the ecosystem services provided by wetlands (Gustavson & Kennedy, 2010).

The economic valuation of wetland ecosystem services can help inform a balanced assessment of the importance of ecosystems for human wellbeing and the economy (Gleason et al., 2008). Wetlands are described as the kidneys of the landscape because of the chemical and hydrological processes they perform (Barbier et al., 1997). Most wetland services are public goods and their consumption is non-excludable. Despite being the only ecosystems with an international treaty calling for their protection (the Ramsar Convention), the degradation of wetlands continues to be exacerbated by ignorance about the values of their ecosystem services and, in particular, that of their non-market environmental services (Ajibola, 2012).

The values of ecosystem services generated by different wetland types are expected to vary. This is certainly true for the services that are closely linked to sediment retention dynamics (Loaiza & Findlay, 2008). However, in most watershed-scale economic valuations of ecosystem services, the same unit value for the water filtration service is assigned to all wetland types (Anielski & Wilson, 2010; Hotte et al., 2009). The purpose of this paper is to present a valuation framework for the filtration services for suspended sediment and phosphorus that explicitly distinguishes between the broad categories of wetlands. The framework is applied to southern Ontario, a region characterized by intensive agriculture that is home to roughly one third of Canada's population. To our knowledge, this is the first study that separately valuates the water filtration functions of different wetland types.

2 | MATERIALS AND METHODS

2.1 | Southern Ontario

The study area comprises the most southerly Mixedwood Plains Ecozone in Ontario, Canada (Figure 1). It is the country's region most affected by human activity (Taylor et al., 2014) and covers 5.33 million hectares, that is, 4.9% of Ontario's total surface. The region experiences high population growth, urban development and intensive farming. Agriculture is presently the dominant land use with natural vegetation reduced to 3% of its historic area. Aquatic ecosystems have deteriorated due to sediment loading and pollution from intensive agriculture, including excess nitrogen (N) and P (Taylor et al., 2014). Wetland area has declined by more than 70% since European settlement (c.1800). Southern Ontario is completely mapped in the Canadian Wetland Inventory (CWI). Based on the Southern Ontario Land Resource Information System (SOLRIS) land use data (MNR, 2008), the areas of the four major wetland type are: bogs (0.85%), fens (0.58%), marshes (11.72%) and swamps (86.85%; Table 2). The total area of all wetlands is 896 149 hectares.

2.2 | Valuation methodology

The water filtration services (i.e., sediment and P retention) are valuated separately for each of the four wetland types by applying the general valuation framework of Turner et al. (2000) illustrated in Figure 2. The sediment and phosphorus accretion rates are used to quantify the water filtration services. We determine the mean sediment and phosphorus accretion rates for each wetland type (see Section 2.2.1) to link the wetland functions and processes with the ecosystem services provided. The wetland value functions are then calculated with Equations (1) and (2):

$$V_{si} = 100 * R_i * A_i * SR_C \tag{1}$$

where V_{si} is the total value (in $ per year) of sediment retention by the i-th wetland type, R_i is the mean sediment accretion rate (cm/year) of the i-th wetland type, A_i is the total surface area of the i-th wetland type in southern Ontario (ha), and SR_C is the sediment removal cost (in $ per m^3); and:

$$V_{pi} = 0.1 * R_i * A_i * D_i * Pr_i * PR_C \tag{2}$$

where V_{pi} is the total value (in $ per year) of phosphorus retention by the i-th wetland type, D_i is the mean soil density (g/cm^3) in the i-th wetland type, Pr_i is the mean phosphorus soil concentration (mg/kg) in the i-th wetland type, and PR_C is the phosphorus removal cost ($/kg).

2.2.1 | Sediment accretion rates

We relied on literature data to estimate representative sediment accretion rates of the different wetland types. The two methods commonly applied to measure sediment accretion data are mass balancing and geochemical tracers. The mass balancing method involves monitoring suspended matter inflow to and outflow from a wetland. Geochemical tracer analysis involves the isotopic dating of sediment cores

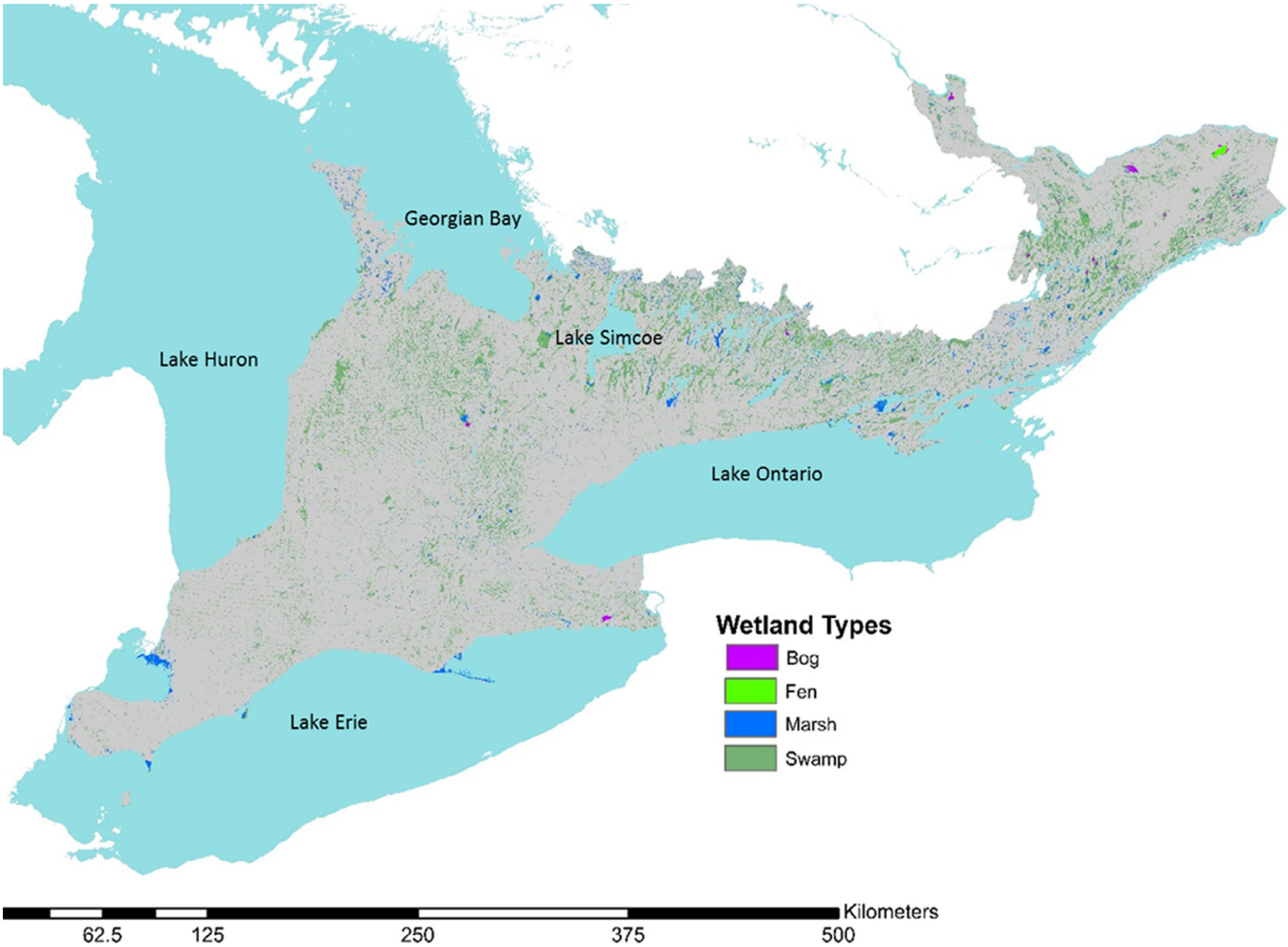

FIGURE 1 Wetland types in southern Ontario, Canada. The area in grey is the selected/study region (MNR, 2008)

TABLE 2 Economic valuation of sediment and phosphorus (P) filtration services by the four major wetland types in southern Ontario

Parameters	Wetland types: Bog	Fen	Marsh	Swamp
Area, A (ha)	7623	5241	104 991	778 294
Area (%)	0.85	0.58	11.72	86.85
Sediment accretion rate (cm/year)	0.23 ± 0.1/0.03 (10[a])	0.14 ± 0.1/0.03 (9[a])	0.36 ± 0.2/0.05 (14[a])	0.22 ± 0.1/0.02 (16[a])
Sediment retention rate (m^3/ha/year)	23 ± 10	14 ± 10	36 ± 20	22 ± 10
P content in soil (mg/kg)	1110 ± 730 (Fennessy et al., 2004)	975 ± 390 (Fennessy et al., 2004)	920 ± 440 (Bruland & Richardson, 2006; Fennessy et al., 2004)	900 ± 370 (Fennessy et al., 2004)
P retention rate (kg/ha/year)	44.7 ± 35	23.9 ± 19	57.9 ± 42	34.6 ± 21
Sediment retention value, Vsi ($/ha/year)	3910 ± 2470	2380 ± 2020	6120 ± 4410	3740 ± 2415
P retention value, Vpi ($/ha/year)	850 ± 885	455 ± 480	1100 ± 1105	660 ± 600
Total sediment retention value ($10^6 \times$ $/year)	30 ± 19	13 ± 11	645 ± 465	2910 ± 1880
Total P retention value ($\times 10^6$ $/year)	6.5 ± 7	2.4 ± 2.5	115 ± 116	513 ± 466

Note: Unit values ($/ha/year) for sediment retention (Vsi) and phosphorus retention (Vpi) are computed with Equations (1) and (2). Average sediment removal costs of $170 ± 78/m^3 and $19 ± 13/kg are used for sediment and phosphorus, respectively. A constant dry soil bulk density of 1.75 mg/cm^3 is assumed for all soils. The soil P concentrations are those reported by Fennessy et al. (2004) based on a meta-analysis of soil chemistry data of Ohio wetlands carried out by the U.S. Environmental Protection Agency. The total sediment retention and total P retention values are then obtained by multiplying the total surface area (A) of each wetland type in southern Ontario with the corresponding unit values for sediment and P retention (Vsi, Vpi). All error estimates in the table are standard deviations (*SDs*), except for sediment accretion rates where standard errors (*SEs*) are also given (value after/symbol).
[a]Sample size.

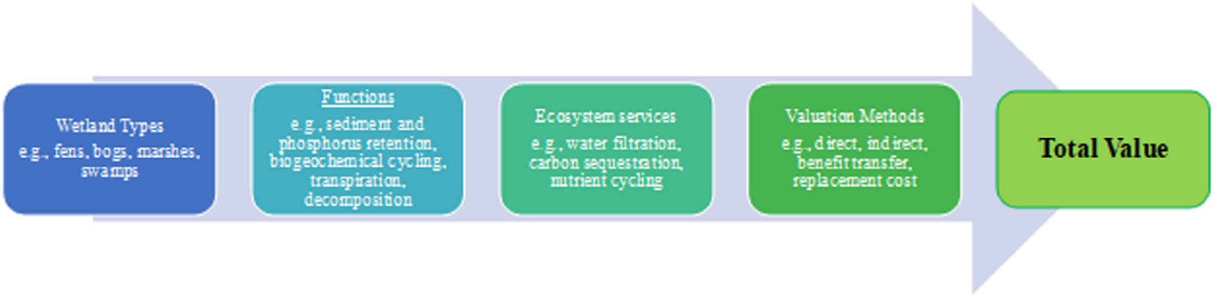

FIGURE 2 A framework for valuation of ecosystem services from different wetland types (modified from Turner et al., 2000)

(Demissie & Fitzpatrick, 1992). In radiometric dating, radionuclides are used as chronological markers. The two natural radionuclides that are most frequently employed are ^{210}Pb and ^{14}C (Church et al., 1987; Walker et al., 2007). Additional artificial radionuclides (^{137}Cs and ^{14}C) released into the environment by nuclear weapon testing and the in situ deployment of geochemical markers are further helping to date sediment and soil sequences in wetlands (Le Roux & Marshall, 2011).

Different measurement methods often yield different rates of sediment accretion. For example, short-term deployments of tracer pads for a few years tend to give higher rates of sediment accretion compared to long term dating methods (e.g., using ^{137}Cs and ^{210}Pb) because short-term measurements do not account for shallow subsidence within the top layer of sediment (Ensign et al., 2014). In our analysis, most of the values listed in Tables A1–A4 were taken from studies that applied long-term measurement techniques (Church et al., 1987; Craft, 2007; Neubauer et al., 2002).

We investigated whether the sediment accretion rates in the different types of wetlands are significantly correlated with the wetland surface area (Figure A1). While we found positive correlations, these ranged statistically from insignificant to weak, in part because of the limited number of rates that could be obtained from the literature. Hence, in the valuation calculations, we assigned a constant sediment accretion rate to each wetland type, which was calculated as the arithmetic mean of the values in Tables A1–A4. Note that the majority of the values used to compute the arithmetic means were taken from studies on wetlands in the United States (see Tables A1–A4).

2.2.2 | Phosphorus retention

The total P concentration in wetland soils typically declines with increasing depth (Craft & Chiang, 2002; Fisher & Reddy, 2010). Below 5–10 cm, the concentration tends to stabilize, indicating that P turnover processes have ceased (Wang et al., 2008). Therefore, we used mean values of the total P concentrations measured on soil samples taken at depth of at least 10 cm as representative for long-term P retention in a wetland (Pinder et al., 2014). The mean total P soil concentrations are then multiplied by the estimated sediment accretion rates to calculate P retention rates (Equation 2).

3 | RESULTS AND DISCUSSION

3.1 | Sediment retention

Most sediment accretion rates fall in the range 0.1–0.5 cm/year (Tables A1–A4 and Figure A1). Fens tend to have the lowest sediment accretion rates, marshes the highest. For each wetland type, a relatively weak positive correlation is observed between the sediment accretion rate and the wetland surface area. The positive correlations may reflect enhanced trapping of sediment in larger wetlands, because of longer hydraulic residence times and more efficient depocenters, similar to reservoirs (Maavara et al., 2015). The average sediment accretion rates decrease in the order marshes > bogs ≈ swamps > fens. For the preliminary valuations presented here, we used the arithmetic mean accretion rate of each of the wetland types to compute the annual sediment and P retentions (Table 2). The mean rates are calculated by averaging the individual accretion rates, which are extracted from the literature and listed in Table A1–A4. Admittedly, the use of a constant mean sediment accretion rate per wetland type is a strong simplification and represents a source of uncertainty in the economic valuation of the filtration functions. Future work should explore in more detail the variability of sediment accretion rates in wetland systems in order to refine the assessment of their role in sediment retention.

3.2 | Phosphorus retention

The average total P concentrations given in Table 2 are mainly those reported by Fennessy et al. (2004) for soil depths of 10 cm in wetlands from Ohio. The latter wetlands are assumed to be reasonable analogues for southern Ontario, as the two agriculture-intensive regions exhibit similar landscapes, climate and cropland P balances (Bruulsema et al., 2011). Sites include forest and shrub vegetation for swamps, depressional, mainstream and headwaters for marshes, and meadows and calcareous wetlands for fens. Concentrations for another 15 marshes within the Painter Creek Watershed in Minnesota (USA) were also included in the calculation of the mean P retention in marshes in Table 2 (Bruland & Richardson, 2006). Reported average dry bulk densities of wetlands in Ontario and Alberta are as follows: 1.49 (bogs), 1.54 (fens), 2.0 (marshes) and 1.57 g/cm^3 (swamps; Redding & Devito, 2005). However, for consistency, we systematically imposed a dry bulk density of 1.75 g/cm^3, because this is the value used by Fennessy et al. (2004) to estimate the total P concentrations shown in Table 2.

Existing studies generally point to the efficient retention of total P by wetlands. For example, a mass balance study of the Hidden Valley wetland, Ontario, found that 50% of total phosphorus is trapped by the wetland (Shane et al., 2001). However, for the same wetland the export of bioavailable P (i.e., dissolved orthophosphate) was 22% higher than the corresponding input. Thus, in-wetland transformation processes can significantly alter the chemical speciation and, hence, the bioavailability of P, not unlike those caused by in-reservoir processes (Van Cappellen & Maavara, 2016).

The average P retention rates used here vary by a factor of three between the lowest (fens) and highest (marshes) values (Table 2). Our average rates fall in the mid-range of values observed in a variety of wetlands (0.1–50 kg P/ha/year; Craft & Casey, 2000; Craft & Richardson, 1993; Dunne & Reddy, 2005; Dunne et al., 2005). A similar range of retention rates of 1–58 kg P/ha/year has been reported for constructed wetlands (CWs; Johannesson et al., 2011), as well as higher values, 50–70 kg P/ha/year, for the Old Woman Creek marsh in the western basin of Lake Erie (Mitsch et al., 1989; Shane et al., 2001). The latter research also concluded that the restoration of one-fourth of the original Old Woman Creek marsh area alone could reduce P loading to the western basin of Lake Erie by 25%–30%.

Overall, P retention rates in wetlands are highly variable across landscapes. Here, relatively high mean values are used because the wetlands of southern Ontario are all located in agricultural watersheds and thus receive high P loads, which in turn results in higher retention rates than for non-agricultural watersheds (Johnston, 1991; Riemersma et al., 2006). The high standard deviations in Table 2 imply that the mean P retention rates yield preliminary, order of magnitude, estimates of the values of the corresponding service.

3.3 | Wetland value functions (V_{si}, V_{pi})

To determine the unit values of the filtration services, we used the average cost for sediment removal and disposal ($SR_C = \$170 \pm 78$ per cubic meter) compiled from data from 10 stormwater management facilities in Ontario (Aziz, 2018). The SR_C estimate thus reflects local practices and costs. Similarly, the total phosphorus removal cost ($PR_C = \$19 \pm 13$ per kg P) is based on the historic performance and costing of 12 water pollution control plants (WPCP), one wastewater treatment centre (WWTC) and a sewage treatment plant (STP), all located in Ontario (Aziz, 2018). Hence, the PR_C estimate also reflects local practices and socio-economic conditions. The use of locally based cost values is key to reducing uncertainties in ecosystem services valuation studies, as opposed to relying on the transfer of values obtained in studies carried out in other locations or context (Aziz & Van Cappellen, 2020). Note that the costs are adjusted using the inflation calculator of Bank of Canada, and expressed in 2016 equivalent Canadian dollars (CAD).

The unit values for sediment and P retention are calculated as the products of the corresponding cost and retention rate values (Table 2). The unit values for sediment and P retention follow the same relative trend as a function of wetland type (Figure 3). The combined unit values ($/ha/year) for the sediment plus P filtration service increase in the order of fens (2835 ± 2075), swamps (4400 ± 2490), bogs (4760 ± 2625) and marshes (6765 ± 4435). The relatively large standard deviations on the unit values are in line with the large ranges in unit values typically reported in ecosystem services valuation studies (Aziz, 2018). Consequently, relative differences in valuation results for given services tend to be more meaningful than absolute differences. Thus, we tentatively conclude that marshes are the most valuable wetlands with respect of their water filtration service, and fens the least.

All our filtration unit values significantly exceed previous estimates. For example, a study of Ontario's Lake Simcoe basin's natural

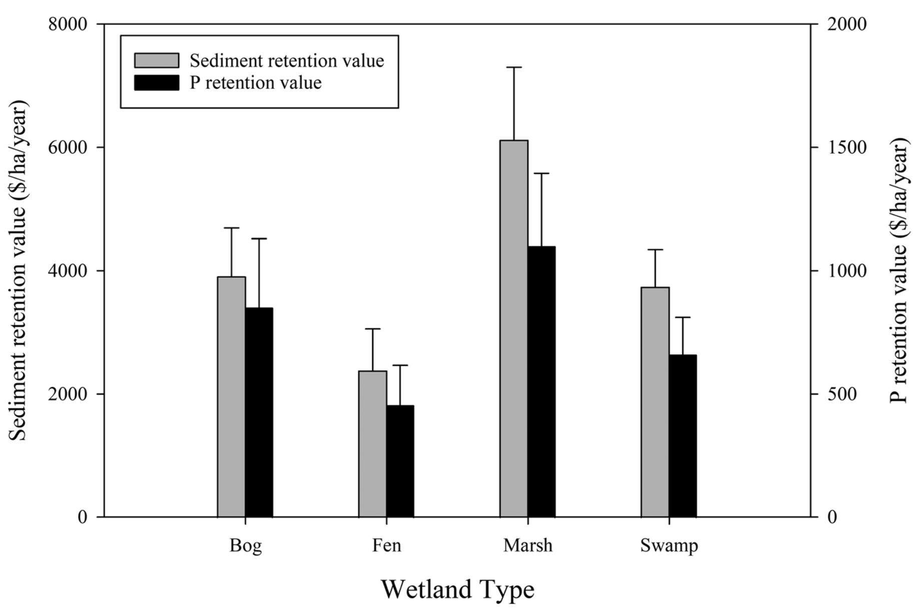

FIGURE 3 Unit values (±*SD*) of sediment and P retention in the four wetland types in southern Ontario (error bars show the standard deviations of the mean values)

capital used a value of $466/ha/year (2016 CAD) for the water filtration service. This value was deduced from a statistical analysis of the potential increase in water treatment costs due to reduction in wetland cover in the United States (Wilson, 2008a). Anielski and Wilson (2009) proposed a very similar water filtration unit value of $452/ha/year (in CAD 2016) across all wetlands types based on a meta-analysis for freshwater wetlands. Another study of ecosystem services in the Greenbelt surrounding the Greater Toronto Area (Wilson, 2008b) assigned a single water filtration unit value of $566/ha/year (CAD 2016). This value was derived from estimates of the potential increase in water treatment costs due to a decrease in forest cover. At the global scale, wetlands have been assigned an even lower value of $259/ha/year (CAD 2016) for their water filtration service (Schuyt & Brander, 2004).

The large discrepancies in unit values between our and other studies illustrates the lack of a unified approach in the valuation of the water filtration service of wetlands, which in turn may cause ambiguities and misunderstandings. These discrepancies emphasize the need to clearly outline the basis of the cost estimates. Our estimates are the highest, because they require that the full capacity to trap sediment and P by the existing wetlands be conserved and accommodated entirely by improved conservation practices and built infrastructure. That is, we valuate the water filtration service of wetlands by matching the original benefits (see also Breaux et al., 1995; Lambert, 2003). Other approaches, including those in the studies mentioned above, estimate the downstream increase in treatment costs that would result from the loss of the existing natural retention capacity. These costs, however, are attenuated by in-stream dilution, retention and transformation processes and therefore only represent a fraction of the value of the lost ecosystem service.

From a sustainable management perspective, the high values of the sediment and phosphorus filtration functions must be assessed in conjunction with the many other ecosystem services provided by wetlands and the interlinkages between these services. In a worst case scenario, a high sediment trapping caused by excessive sediment loading to a wetland may result in ecological degradation, for example by causing habitat instability and loss (Sileshi et al., 2020). Similarly, a high phosphorus filtration efficiency can lead to the undesirable eutrophication of a wetland. In the long run, these negative impacts may even cause a reduction in the sediment and phosphorus filtration functions themselves. Thus, when using the estimated values of the filtration functions to inform environmental decision making, the finite filtration capacities and resilience of the affected ecosystems need to be considered.

3.4 | Total wetland filtration service value

The unit values for the water filtration service provided by each wetland type are applied to the respective wetland areas in southern Ontario to obtain the total values of phosphorus and sediment retention by all wetlands (Table 2). These total values are strongly dependent on the relative surface areas covered by the different types of wetlands in southern Ontario. For instance, even though the unit values of swamps are approximately half those of marshes, swamps dominate the total value because they make up most (87%) of the total wetland area in the region (Figure 4). The total value of water filtration service (sediment plus P removal) performed by all wetland types in southern Ontario amounts to $4.2 ± 2.9 billion per year (CAD 2016). Furthermore, the value of sediment retention by wetlands is about six times higher than that of phosphorus retention.

3.5 | Offsetting P retention by existing wetlands

Phosphorus is the ultimate limiting nutrient in streams and lakes in and around southern Ontario (Schindler, 2012). The only method that has so far proven successful in controlling eutrophication of the

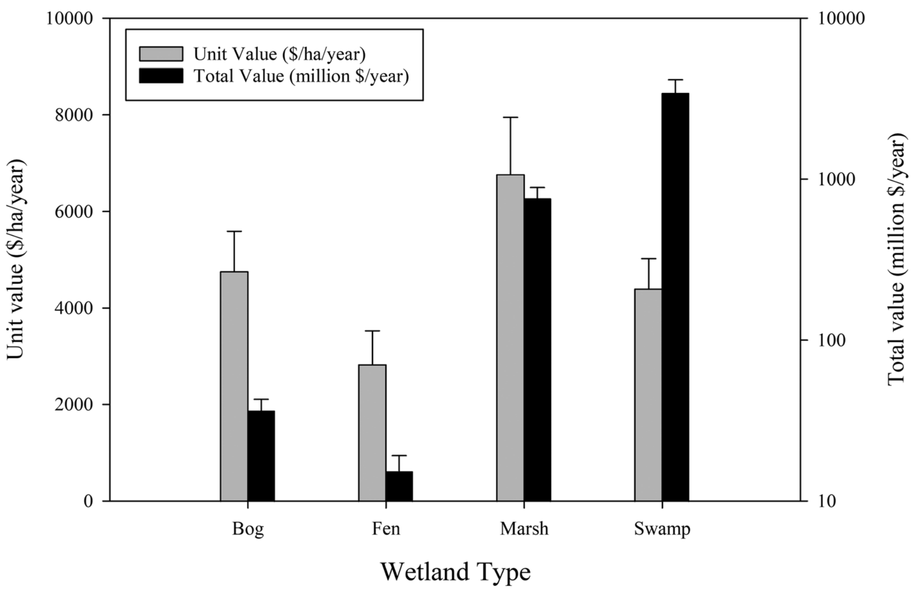

FIGURE 4 Total value of sediment and phosphorus removal from water for the four wetland types in southern Ontario (error bars show standard deviations of the mean values)

TABLE 3 Costs of three interventions—Best management practices (BMPs), constructed wetlands (CWs), waste water treatment plants upgrades (WWTPUs)—To offset P released from the loss of 1 ha of wetland from the four types, as well as from the loss of all existing wetland area

Alternatives	Cost ($\times 10^3$) $/year to offset excess P from loss of 1 ha of wetland				Cost (billion $/year) to offset excess P from loss of all wetlands
	Bog	Fen	Marsh	Swamp	
BMPs	19.2 ± 8	8.7 ± 6	27.9 ± 17	13.2 ± 7	13.40 ± 5
CWs	4.1 ± 2.8	1.9 ± 1.7	6.0 ± 4.9	2.8 ± 2.1	2.90 ± 1.2
WWTPUs	236.0 ± 102	106.8 ± 83	342.2 ± 213	1611.5 ± 84	164.0 ± 69

Note: The total existing wetland area retains 29 944 ± 12 659 t P/year.

region's lakes, in particular Lake Erie and Lake Ontario, is to reduce P inputs (Schindler, 2012). To address the resurgence of algal blooms in Lake Erie, the United States and Canada have committed to reduce P inputs to the lake by 40% from the year 2008 baseline, which means an annual reduction of 200 metric tonnes of P from the Canadian side (Hanief & Laursen, 2019).

For the cost-effectiveness analyses, we assessed the cost of replacing wetlands' P retention capacity under a scenario where all existing wetlands are converted to agriculture. Using the P retention rates of the four wetland types and their areas, the total annual P retention by wetlands in southern Ontario is close to 30 000 tonnes (Table 2). The additional P load from converting wetlands to agricultural land was calculated using the average P export rates for row crops, small grains, forage and pasture from local and regional studies (Donahue, 2013; Jeje, 2006; Shaver et al., 1994; Winter, 1998). Using an estimated composite delivery rate of 0.52 ± 0.28 kg P ha^{-1} $year^{-1}$, the additional P load is then 466 ± 251 t P/year. Therefore, the total P loading from wetland loss and additional agricultural P is 30420 ± 11 990 t P/year. We now consider three alternatives to offset this excess P load: (1) best management practices (BMPs), (2) CWs and (3) wastewater treatment plant (WWTP) upgrades. The cost of converting 2 ha of each wetland type plus that of converting all wetlands to agriculture is estimated using these three alternatives (Table 3).

3.5.1 | Best management practices

A generally accepted cost for removing 1 kg P by completed BMPs projects in southern Ontario is $400/year (CAD 2009). This includes the cost of the BMP implementation and project management (Marcano, 2015). When accounting for inflation, the value in 2016 is $447 per kg of P removal per year. The annual total cost of offsetting the lost P retention via BMPs then equals about $13 billion (Table 3).

3.5.2 | Constructed wetlands

Kynkäänniemi et al. (2013) report that newly constructed wetlands retain 69 ± 36 kg/ha/year of total phosphorus TP, based on 2 years of operation. Using this retention rate, the area of constructed wetlands required to offset the increased P load is 440846 ± 288 255 ha, or about 50% of the existing (natural) wetland area. The annual cost of operating a functional wetland of size 1.125 ha in Embrun, eastern Ontario, with an estimated lifespan of 30 years, is $7420 (CAD 2016). This cost is based on interest on capital investment, operation and maintenance cost, annual depreciation and loss of crop yield on the land (Tousignant et al., 1999). Scaling the cost to the entire area of constructed wetlands required then yields a total cost of $2.9 billion (Table 3).

3.5.3 | WWTPs upgrades

A cost–benefit analysis of phosphorus in the Grand River watershed, Ontario, suggests that, if all the WWTPs are upgraded in the watershed, it will cost $5475 to remove 1 kg of P (CAD 2016; Hanna, 2015). This cost does not include the optimization of operation of current processes in the upgrading option. Using this cost, WWTPs become the most expensive option to offset the lost P from conversion of wetlands to agriculture: $164 billion per year (Table 3).

The results in Table 3 indicate that the options for phosphorus removal considered are not cost effective when compared to the P retention service values of the existing wetlands in Table 2. The least expensive option is constructed wetlands, however it requires that land is made available to install the new wetlands. The areas of constructed wetlands required to counteract the extra P loads generated by the loss of 1 ha of bog, fen, marsh and swamp are 0.62, 0.28, 0.41 and 0.91 ha, respectively. The required area of constructed wetlands is almost equal in the case of marshes because of the very similar P retention rates.

4 | CONCLUSIONS

This study presents a first valuation of the sediment and P water filtration services of wetlands in southern Ontario. The estimates are based on mean sediment accretion rates for different wetland types as the master variable regulating the water filtration efficiency for suspended sediment and P. The unit values of the water filtration services of the four wetland types in southern Ontario increase in the order: marsh > bog $\approx$ swamp > fen. Hence, marshes are the most valuable wetland type for water filtration. Our cost-effectiveness analysis further shows that it would be very costly to replace the existing wetlands' water filtration services by improved land and nutrient management and manmade infrastructure. Further work should refine the valuation estimates presented here by more precisely delineating the relationships between wetland size and sediment accretion rates, and by accounting for the hydrological connectivity of wetlands across the landscape as well as the variability of concentration, speciation and mobility of sedimentary phosphorus. In addition, the filtration functions assessed here are part of a much larger set of ecosystem services provided by wetlands.

DATA AVAILABILITY STATEMENT

Data sharing is not applicable to this article as no new data were created or analyzed in this study.

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APPENDIX A.

The sediment accretion rates summarized in Tables A1–A4 are plotted against the wetland size in Figure A1. The observed scatter reflects differences in site-specific characteristics within each wetland type (e.g., surrounding land use, recharge source water, discharge pathways, wetland morphology and vegetation). In the data set most of the fens are less than 10 ha in area, the bogs less than 100 ha. Marshes and swamps are the only wetland types that exceed 100 ha in size in southern Ontario. The frequency distributions of the sediment accretion rates are shown in Figure A2. Because of the absence of clear distribution patterns and limited data, the arithmetic mean value for each wetland type is used in the valuation calculations.

TABLE A1 Sediment accretion rates for bogs

Bog name	Area	Accretion rate (cm/year)	Reference
Wylde Lake Bog	460.72 ha (cwi)	0.059 ± 0.001a	Shiller (2013)
Marcell S-2 Bog	3.2 ha	0.24	Wieder et al. (1994)
Big Run Bog	15 ha	0.31	Wieder et al. (1994)
Tub Run Bog	23 ha	0.23	Wieder et al. (1994)
Cranberry Bog 1	65 ha	0.055	Kadlec and Robbins (1984)
Cranberry Bog 2	65 ha	0.23	Kadlec and Robbins (1984)
Alfred Bog	4000	0.05a	Bird and Hale Limited (1984)
Burns Bog	4000 ha	0.42	Biggs (1976)
Sifton Bog	41.6 ha	0.18	Le Roux and Marshall (2011)
Mer Bleue Bog	2800 ha	0.21	Talbot et al. (2010)

Note: Wylde Lake Bog is located in Luther marsh, Grand River watershed and area is obtained from Canadian wetland inventory (CWI).
[a]Results are calculated for long time period (more than 300 years) and are not used in our analysis.

TABLE A2 Sediment accretion rates for fens

Fen name	Area	Accretion rate (cm/year)	Reference
Drosera Fen, Yosemite National Park	5.03 ha	0.39 ± 0.15	Drexler et al. (2015)
Porcupine Fen, Yosemite National Park	0.98 ha	0.16 ± 0.02	Drexler et al. (2015)
Kiln Fen, Sagehen basin	2.2 ha	0.08 ± 0.04	Bartolome et al. (1990)
Two field East Fen	0.8 ha	0.05 ± 0.009	Bartolome et al. (1990)
West Fen	0.1 ha	0.03 ± 0.02	Bartolome et al. (1990)
Bagno Bruch	39 ha	0.13	Fia kiewicz-Kozie et al. (2014)
Bagno Mikołeska	5 ha	0.16	Fia kiewicz-Kozie et al. (2014)
Abeille fen	3.5	0.15	Van Bellen et al. (2013)
LG1 fen, Quebec	20	0.12	Beaulieu-Audy et al. (2009)

TABLE A3 Sediment accretion rates for marshes

Marsh name	Area (ha)	Accretion rate (cm/year)	Reference
Hank's marsh	438.84	0.28 ± 0.03[a]	Graham et al. (2005)
Upper Klamath NWR	3484	0.54	Graham et al. (2005)
Squaw Point	133	0.42 ± 0.03	Graham et al. (2005)
Corte Madera Marsh	121	0.4 ± 0.07	Callaway et al. (2013)
Barataria basin marsh	4780	0.65	Hatton et al. (1983)
Dyke Marsh	37.5	0.31	Elmore et al. (2015)
Sweet Hall marsh	401	0.53 ± 0.11	Neubauer et al. (2002)
Great Marsh, Delaware	6880[b]	0.5	Church et al. (1987)
Ogeechee marsh, Georgia, USA	700	0.21	Loomis and Craft (2010)
Altamaha marsh	3700	0.12	Loomis and Craft (2010)
Satilla marsh	1700	0.23	Loomis and Craft (2010)
Jug Bay marsh Maryland	607[c]	0.5	Khan and Brush (1994)
Gleason marsh	85[c]	0.27	Darke and Megonigal (2003)
Walkerton marsh	16[c]	0.12	Darke and Megonigal (2003)

[a]Average of ^{210}Pb and ^{137}Cs models.
[b]Area taken from US National Wetland Inventory.
[c]http://dnr2.maryland.gov/wildlife/Documents/NaturalAreas/JugBay.pdf (Department of Natural Resources Maryland).

TABLE A4 Sediment accretion rates for swamps

Swamp name	Area	Accretion rate (cm/year)	Reference
Tamarack swamp	1618 ha	0.14	Wieder et al. (1994)
Cranesville Swamp	809 ha	0.19	Wieder et al. (1994)
Black swamp Arkansas	1804 ha[a]	0.28	Hupp and Morris (1990)
Walden swamp	26 ha[a]	1.26	Meadowlands Environmental Research Institute (2011)
Eight Day swamp	7.85 ha[a]	0.83	Meadowlands Environmental Research Institute (2011)
Backswamp, Alabama	1163 ha[a]	0.5 ± 0.1 cm/year	Kidd et al. (2015)
Okefenokee Swamp		0.08 cm/year	Craft et al. (2008)
Louisiana swamp		0.49 ± 0.11	Conner and Day (1991)
Bluebonnet swamp	42	0.41	Sanders (1998)
Heron Pond swamp	30	0.8	Warren (2001)
Pointe au Chene swamp	231	0.4	Rybczyk et al. (1998)
Buttonland swamp	1600	0.25	Demissie and Fitzpatrick (1992)
La Union swamp	10	0.052	Urquhart (1999)
Tuckean Swamp	5000	0.22	Taffs and Heijnis (2008)
Nariva Swamp	6234[b]	0.31	Ramcharan (2004)
Loboi Swamp	150	0.1	Ashley et al. (2004)

[a]Area from U.S. National Wetland Inventory.
[b]http://www.ema.co.tt/new/images/guides/AppendixB.pdf.

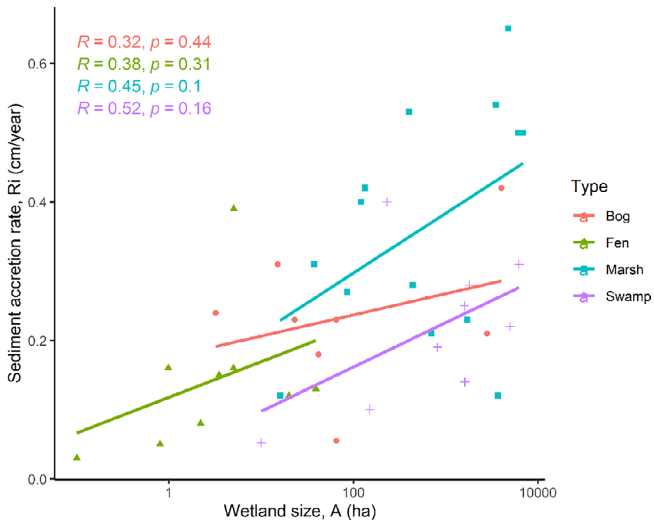

FIGURE A1 Sediment accretion rates (cm/year) versus wetland size (as surface area in ha) for the four wetland types

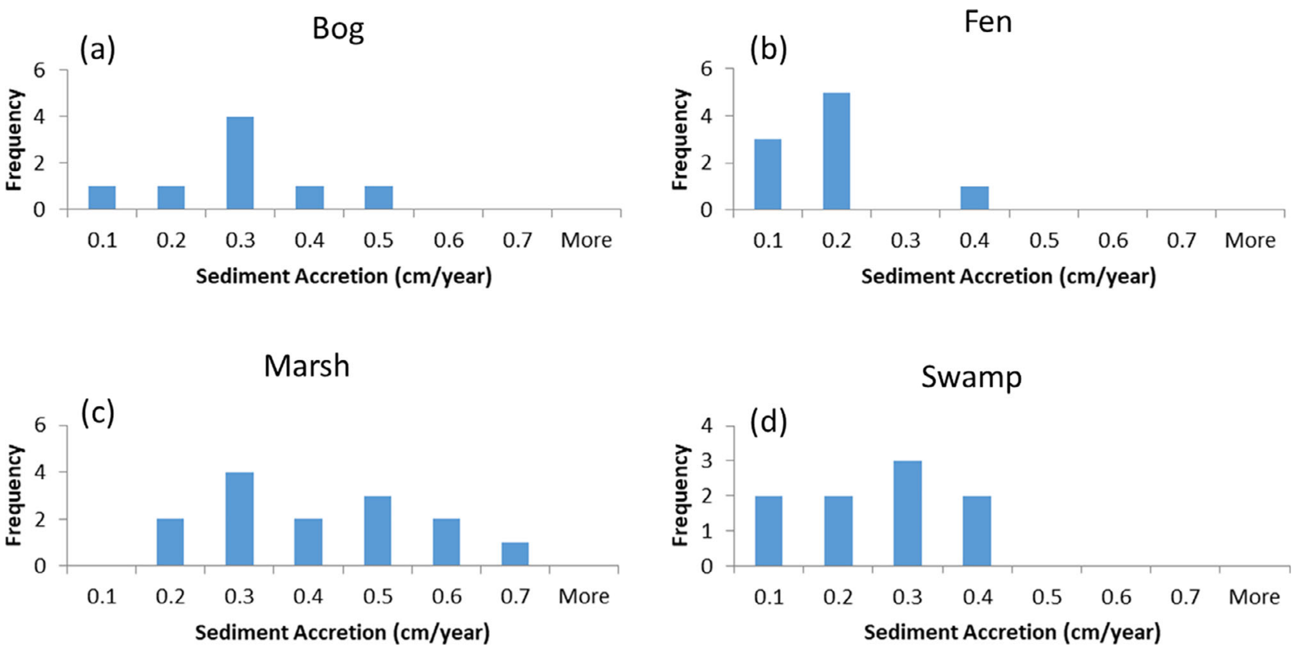

FIGURE A2 Frequency distributions of the sediment accretion rate data for the four wetland types. The data are given in Tables A1–A4