Global Ecology and Conservation 2 (2014) 11–23

Contents lists available at ScienceDirect

Global Ecology and Conservation

journal homepage: www.elsevier.com/locate/gecco

Original research article

Towards systematic conservation planning adapted to the local flow of ecosystem services

Jérôme Cimon-Morin[a,b,c,*], Marcel Darveau[b,a,c], Monique Poulin[a,c]

[a] Laval University, Pavillon Paul-Comtois, Faculté des sciences de l'agriculture et de l'alimentation, Département de phytologie, 2425 rue de l'Agriculture, Québec, Qc, Canada G1V 0A6

[b] Ducks Unlimited Canada, 710 Bouvier, bureau 260, Québec, Qc, Canada G2J 1C2

[c] Quebec Centre for Biodiversity Science, Montreal, McGill University, Stewart Biology Building, Department of Biology, 1205 Dr. Penfield Avenue, Montreal, Qc, Canada H3A 1B1

ARTICLE INFO

Article history:
Received 10 July 2014
Received in revised form 15 July 2014
Accepted 15 July 2014
Available online 30 July 2014

Keywords:
Ecological services
Priority areas
Wetlands
Demand
Conservation assessments
Beneficiaries

ABSTRACT

Ecosystem services (ES) are increasingly included in conservation assessment worldwide to sustain their ability to fulfill human needs. Due to the instrumental value inherent in ES, priority areas for their conservation should be selected based on their capacity to both ensure an available supply and meet beneficiary demands. However, such a methodology has yet to be developed. Aiming to adapt systematic conservation planning procedures to include ES, we conducted a case study in eastern Canada focusing on ten ES for 16 wetland types. We first delimited the ES supply accessible for human use from the total biophysical supply and mapped demand for each ES. Secondly, we assembled conservation networks targeting the accessible supply and demand and compared them with networks targeting either ES biophysical supply or accessible supply. We found that targeting only ES supply resulted in selecting sites that are not in demand and may be up to three times less efficient in fulfilling the demands of beneficiaries for local flow ES. Thus, not considering demand in ES conservation assessment fails to position reserves where ES supply is likely to be most useful. Setting conservation targets for ES supply and demand could therefore help to achieve ES conservation objectives.

1. Introduction

Steady expansion of the world's population and economic growth will continue to increase pressure on natural ecosystems and accelerate the decline of the supply of most ecosystem services (ES) observed around the globe (Chapin et al., 2000; Foley et al., 2005; Millennium Ecosystem Assessment (MA), 2005; Vitousek et al., 1997). ES have been defined as the benefits that humans obtain from ecosystems and have been classified according to four categories: provisioning, regulating, supporting and cultural services (Millennium Ecosystem Assessment (MA), 2005). In the short term, modern land use practices can increase the supply of most provisioning services (i.e. food and material), but in the long term they undermine the capacity of ecosystems to provide other services, such as freshwater supply, climate regulation and recreational opportunities (Foley et al., 2005; Millennium Ecosystem Assessment (MA), 2005). The growing awareness of the importance of ES for human well-being has increased interests in securing their sustainability, notably through land protection and related

* Corresponding author at: Laval University, Pavillon Paul-Comtois, Faculté des sciences de l'agriculture et de l'alimentation, Département de phytologie, 2425 rue de l'Agriculture, Québec, Qc, Canada, G1V 0A6. Tel.: +1 418 623 1650x27.
E-mail address: jerome.cimon-morin.1@ulaval.ca (J. Cimon-Morin).

http://dx.doi.org/10.1016/j.gecco.2014.07.005
2351-9894/

conservation actions (Balvanera et al., 2006; Chan et al., 2006; Egoh et al., 2007; Millennium Ecosystem Assessment (MA), 2005; Turner et al., 2007). Human societies' demand for and dependence on ES is expected to grow (Guo et al., 2010), and along with it, the need to sustain ES availability.

ES provide benefits on different spatial flow scales (i.e. ranging from local to global), depending on where a service is produced (source) relative to where its benefits can be perceived (sink) by human beneficiaries (Bagstad et al., 2013; Balmford et al., 2011; Cimon-Morin et al., 2013; Fisher et al., 2009). Protected areas for ES have to be identified based on their capacity to provide a continuous flow of ES to their specific beneficiaries. From a conservation perspective, most ES have a local spatial flow scale; for this reason beneficiaries must approach or enter the protected area where the ES are supplied to obtain its benefits (thereafter referred as "local flow ES"). For example, recreational angling in a protected area requires the angler to capture (sink) the fish species within the protected area (source), established to conserve nature and its associated ES, even if the benefit (i.e. the meat) can be consumed elsewhere. Moreover, demand for protecting ES, or the sum of the benefits currently obtained in a particular area (Burkhard et al., 2012), is spatially heterogeneous (Burkhard et al., 2012; Nedkov and Burkhard, 2012; van Jaarsveld et al., 2005). Demand for local flow ES generally diminishes with increasing distance from beneficiaries because far fewer people are willing to travel great distances to obtain benefits from nature (Chan et al., 2006; Holland et al., 2011). A spatial mismatch can thus occur between local flow ES supply (i.e. the amount of benefits) and the sites most used by human beneficiaries (i.e. highest demand). For example, demand for recreation services is driven more by the proximity to roads and the size of and the distance to nearby population centers than by the capacity of a site to provide the services per se (Chan et al., 2006; Holland et al., 2011). Accordingly, local flow ES do not necessarily provide actual benefits to human populations everywhere they are supplied, either due to lack of physical access or demand or restrictions by institutional arrangement (e.g. land-use constraints in national parks restrict access to provisioning services; Tallis et al., 2012).

Systematic conservation planning (SCP) is increasingly recommended for safeguarding ES provision (Chan et al., 2006; Cimon-Morin et al., 2013; Egoh et al., 2008). SCP is a multi-component stage-wise approach to identifying conservation areas and devising management policy, with feedback, revision, and reiteration, where needed (Kukkala and Moilanen, 2013; Margules and Sarkar, 2007; Pressey and Bottrill, 2008; Sarkar and Illoldi-Rangel, 2010). SCP notably involves identifying priority areas to effectively achieve conservation goals; traditionally, these goals include representativeness, persistence and cost-efficiency (Margules and Sarkar, 2007). However, due to the anthropocentric focus and instrumental value associated with ES (Reyers et al., 2012), these goals must be expanded to address the spatial relationships between ES supply and their human beneficiaries (Chan et al., 2006; Egoh et al., 2007). Specifically, ES conservation areas should be targeted as a complementary set of sites selected according to their capacity to ensure a sustainable and accessible supply of ES as well as deliver these benefits where they are needed (Cimon-Morin et al., 2013).

Although an increasing number of studies have included ES in conservation assessments (Chan et al., 2006; Egoh et al., 2008, 2007; Larsen et al., 2011; Luck et al., 2009; Naidoo et al., 2008), there is still a knowledge gap on how to effectively prioritize areas based on ES provision, accessibility to beneficiaries and demand (Cimon-Morin et al., 2013; Egoh et al., 2007; Maes et al., 2012; Tallis and Polasky, 2009). The aim of this study is therefore to suggest a modification of SCP procedures that would increase the effectiveness of local flow ES conservation. For this purpose, we conducted a case study in eastern Canada focusing on 16 wetland and aquatic habitats and an associated set of 10 ES (five provisioning, three cultural and two regulating services). We first mapped for each planning unit the biophysical supply of each ES and then used proxies of human occupancy of the territory to define the supply's potential-use spatial range, that is to say, the supply accessible for human use. Concurrently, we mapped ES demand as the probability that a planning unit would be used by beneficiaries in order to obtain the benefits of a specific ES. We compared conservation networks resulting from site-selection algorithms based on the biophysical supply of ES, the potential-use supply or the combination of potential-use supply and demand (i.e. the actual-use supply). The concept of actual-use supply originates from the assumption that the real contribution to human well-being is not only when ES are supplied and the benefits are accessible but also when a minimal amount of demand is fulfilled. Accordingly, the actual-use supply of an ES is defined as when both accessible supply (i.e. potential use supply) and demand occur at the same site. We hypothesized that prioritizing areas based on actual-use supply would foster conservation choices more efficiently towards ES conservation objectives. Finally, we evaluated how to best integrate data on ES demand in SCP to assemble conservation networks that are the most appropriate for satisfying the needs of beneficiaries.

2. Method

2.1. Study area and wetlands mapping

The study was undertaken in the Lower North-Shore Plateau ecoregion and in a southern portion of the Central Labrador ecoregion of boreal eastern Canada (Fig. 1; Li and Ducruc, 1999). The study area covers over 137 565 km^2, most of it part of the black spruce-moss vegetation domain (Saucier et al., 2009). Of the approximately 12 350 inhabitants (0.09 inhabitants/km^2), 9800 are dispersed among fifteen municipalities and 2550 in four First Nations communities (Gouvernement du Québec, 2013). The minimal mapping unit of the Natural-Capital Inventory dataset (Ducruc, 1985), a dataset originally built for the ecological classification of the territory, was used to divide the study area into 16 026 planning units. These units are irregular in shape and size (mean of 8.5 ± 15 km^2) because they are delimited by significant and permanent environmental features, such as landscape topography, surface deposits and water bodies. All mapping was performed using ArcGIS 10.0. (ESRI, 2012). The study area is currently minimally developed but its large freshwater reserves, commercial forests and rivers

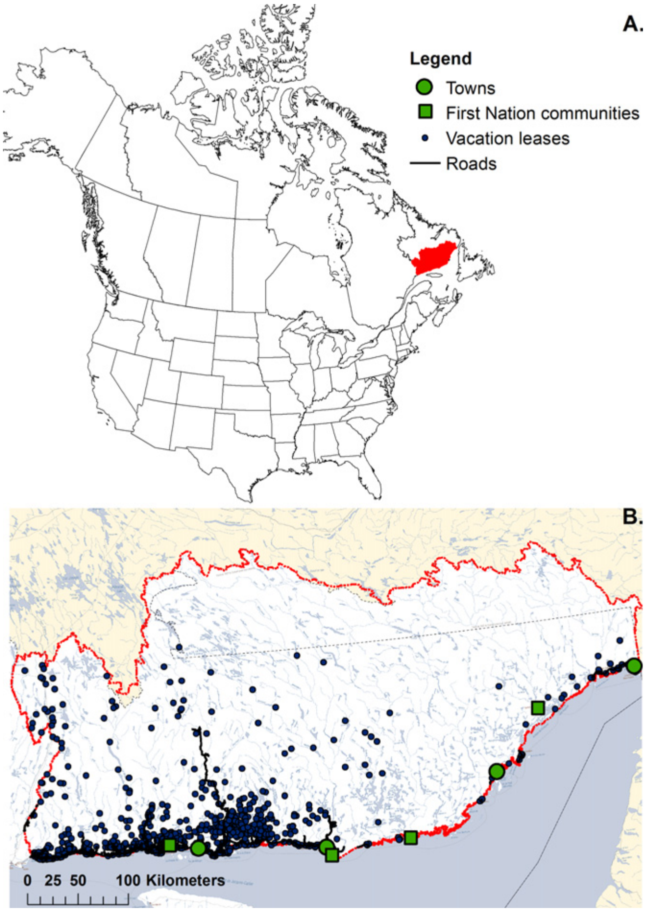

Fig. 1. The location of the study area (colored area) across North America (A); the extent of road networks and the location of the major towns, First Nations communities and vacation leases are shown in (B). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

which are great potential sources of hydroelectricity, as well as the presence of important mineral deposits makes it a great candidate for future industrial development (Berteaux, 2013).

Assuming that the various wetland and aquatic habitat types differ in their capacity to supply ES, it was decided to map 16, the largest number possible using the best available complete data. We used the Natural-Capital Inventory dataset (Ducruc, 1985), which contains aggregated information on descriptive variables at the planning unit scale, such as surface deposit (e.g. organic or mineral), drainage and vegetation cover, to infer the relative coverage proportion of 10 peatland and one mineral wetland types (including marshes and swamps). We differentiated four types of ombrotrophic peatlands (bogs) based on the presence of ombrotrophic organic deposit and using peat depth (thick or thin; threshold of ± 1 m) and vegetation cover (forested or not) attributes. We also discriminated six minerotrophic peatlands (fens) among the minerotrophic organic deposits using peat depth (thick or thin; threshold of ± 1 m) and vegetation cover (forested or not and also presence/absence of strings) attributes. Four types of aquatic habitats (streams, rivers, ponds and lakes) were extracted from the CanVec v8.0 dataset (Natural Resources Canada (NRC), 2011). Lakes were further divided into shallow (littoral zones, <2 m deep) and deep water zones (pelagic zones) using a 100 m distance buffer from the shoreline (Lemelin and Darveau, 2008). This division was based on the premise that these two types of habitats differ in their capacity to generate ES supply, notably for waterfowl related ES (Lemelin et al., 2010). Aquatic habitats were converted into relative coverage proportion for each planning units. Freshwater wetlands, mostly peatlands and shallow waters, cover 10% of the study area, while another 10% is composed of deep freshwater (>2 m deep).

2.2. Mapping ecosystem services supply and demand

For the purpose of this study, we selected five provisioning, two regulating and three cultural services provided by wetlands for which the sustainability of supply is important for tourism and local communities. Regulating and cultural ES are generally compatible with most protected area categories and especially strict conservation status (e.g. IUCN I–III status; Dudley, 2008). Provisioning services can also be included in conservation if there are restrictions on practices (low land-use

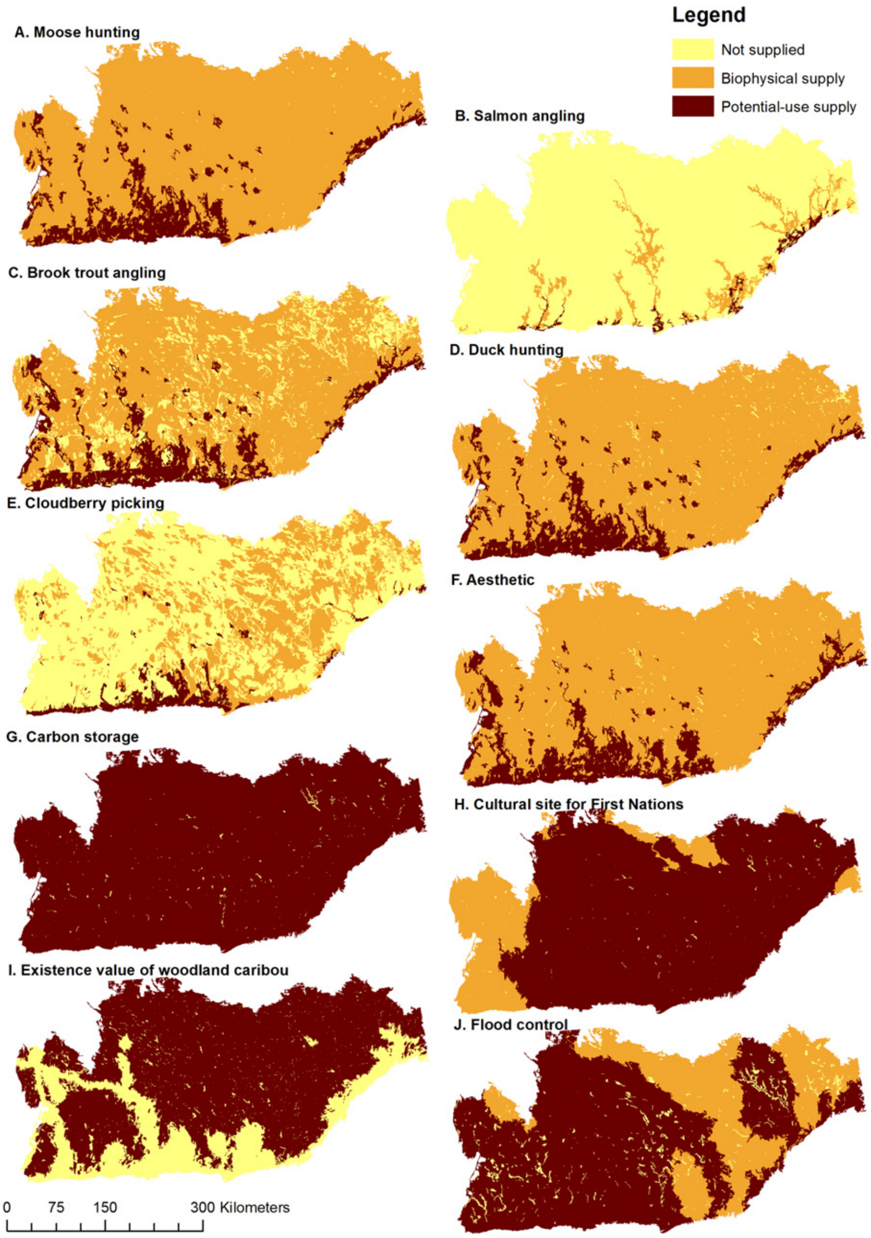

Fig. 2. The spatial delivery range of the biophysical supply and of the potential-use supply of ES. The biophysical supply area represents the zones where the ES is supplied but not necessarily accessible for consumption. The potential-use area illustrates the zones where the ES is supplied and potentially accessible for consumption. The potential-use area is a subset of the biophysical supply area. The not-supplied area shows the zone where the ES is not produced.

intensity; e.g. recreational angling) to ensure sustainability and preclude biodiversity loss (Cimon-Morin et al., 2013). The protection of provisioning ES should require the use of protected area categories that allow some resource extraction for local use while also excluding industrial activity (e.g. IUCN IV–VI status; Dudley, 2008). For each ES, supply and demand were mapped quantitatively (Table 1, see Appendix A for detailed description). Experts were consulted for the quantitative assessment and to validate the mapping of each ES supply and demand.

ES supply was first mapped according to biophysical supply (BS), also known as natural capital. Assuming that any ES can be protected wherever it is supplied, this mapping approach only takes into account the biophysical capacity of wetland types to provide an ES in each planning unit (Table 1 and Fig. 2). Secondly, ES were mapped with regard to potential-use supply (PUS). In other words, the spatial flow scale of each ES and proxies of human occupancy were used to identify the set of planning units in which humans can potentially perceive ES benefits (Table 1 and Fig. 2). Among the ten ES chosen for this study, seven have a local, one a regional and two have a global flow scale. To identify the set of planning units providing

Table 1
Indicators and data used to map ES supply and demand across the study area. See Appendix A for a detailed description.

ES	Spatial flow scale	Indicators used to map ES supply		PUS supply/BS supply ratio	Indicators used to map ES demand
		Biophysical supply (BS)	Potential-use supply (PUS)		
Moose hunting	Local	Composition of planning units in aquatic and wetland moose habitats (Timmermann and McNicol, 1988; Hydro-Québec, 2007; Tecsult, 2006; Lamontagne and Lefort, 2004)	PUS obtained using accessibility proxies[b]	13%	Number of moose hunted per planning unit between 1991 and 2011 (Ministry of Natural Resources, Personal commun.)
Salmon angling	Local	Salmon river layer (Ministry of Natural Resources, Personal commun.), mean number of salmon migrating upstream per river per year (Caron et al., 2006), zones permitting salmon fishing (MRNF, 2012b)	PUS obtained using accessibility proxies	25%	Mean number of salmon fished per river between 2008 and 2012 (MRNF, 2012a) and proxies of human demand for salmon angling[b]
Brook trout angling	Local	Composition of planning in aquatic and wetland trout habitats (Hydro-Québec, 2007), inaccessible water body by fish layer (Bellavance and Gagné, 2012)	PUS obtained using accessibility proxies	19%	Proxies of human demand for brook trout angling
Black duck hunting	Local	Habitat selection ratio of waterfowl (Lemelin et al., 2010; Guérette Montminy et al., 2009; Lemelin et al., 2004)	PUS obtained using accessibility proxies	18%	Proxies of human demand for duck hunting
Cloudberry picking	Local	Fruit yield per wetlands type (C. Naess, Personal commun.)	PUS obtained using accessibility proxies	27%	Proxies of human demand for cloudberry picking
Aesthetics	Local–proximal[a]	Wetland and aquatic habitats composition and heterogeneity (Pâquet, 1997)	Distance buffer of 500 m from all human infrastructure (Pâquet, 2003; Pâquet and Bélanger, 1998)	12%	Demand was estimated according to: (1) the appeal of human infrastructures, (2) mean duration of users' frequentation, (3) and observation, (4) users' expectations and (4) the number of users per planning unit (Pâquet, 2003)
Cultural site for First Nations subsistence uptake	Local	Wetland and aquatic habitats composition of harvested species (Charest, 1996; Walsh, 2005)	Zones actually used by First Nations (Charest, 2005)	86%	Delimitation of high and low uptake zone (Charest, 2005) and uptake intensity in each zone (Walsh, 2005)
Existence value of woodland caribou	Global	Mean probability of occurrence per planning unit (Environment Canada, 2008), buffer zones of avoidance from human disturbances (Dyer et al., 2001; Fortin et al., 2013; Seip et al., 2007; Vistnes and Nellemann, 2007, 2008; Vors et al., 2007)	PUS equals the BS	100%	Demand was set equal across the spatial range of the PUS
Flood control	Regional	The capacity of each planning unit to reduce and stabilize the water that flows through it (Gouvernement du Québec, 1993)	PUS mapped only in watersheds containing human infrastructure	66%	Demand was set equal across the spatial range of the PUS
Carbon storage	Global	Carbon stock value for bogs (Magnan et al., 2011 and personal commun.), for fens (Tarnocai and Lacelle, 1996), for mineral wetlands (Horwath, 2007) were used, carbon stock for lakes and ponds were modeled using the equation provided by Ferland et al. (2012)	PUS equals the BS	100%	Demand was set equal across the spatial range of PUS

[a] Local proximal means that the benefits of this ES can be perceived not only at the point of production, but also at a certain distance from where it is supplied.
[b] See method, Section 2.2.

potential-use benefits to people, the following proxies of accessibility and of human occupancy were used for local flow ES (except for cultural site ES; see below): (1) a 1 km buffer zone around all types of roads and human settlements, such as leases of vacation lots on public lands (mostly used for fishing- and hunting-related activities), and (2) the area occupied by outfitters offering the targeted ES. While these proxies may be a conservative estimate of planning unit accessibility, we believe that the majority of human uses for the targeted local flow ES will take place within these limits. Therefore, planning units that fall outside the spatial range of benefit delivery of an individual ES were considered to provide no accessible benefits and were not considered for the conservation of this ES' supply (i.e. the planning unit feature value was set to nil). For the sole regional flow scale ES, that is, flood control, only the planning units present in watersheds containing human infrastructures were retained in the PUS. For the two global flow ES, the BS and the PUS were identical.

Although the ES supply of a planning unit may be accessible to humans, the benefits are not necessarily in real demand. In order to identify the planning unit providing actual-use benefits (AU), we mapped demand as the probability of a planning unit to be used by beneficiaries in order to obtain the benefits of a specific ES across its potential-use supply spatial range. Demand for global flow ES was considered equal across their PUS range. For regional flow ES (i.e. flood control), the demand may vary according to human population density and the presence of human infrastructures (e.g. roads, bridges, etc.). We were not able to establish precise demand values for each watershed. For the purpose of this study, we assumed that demand for regional flow ES is also equal across the spatial range of their PUS. This raises the need to develop methods for estimating complex spatial demand values for ES. Demand for most local flow scale ES often involves the movement of their beneficiaries, who must go to where the ES is supplied in order to benefit from it. For moose hunting and salmon angling, primary data about demand was available. Demand for the other local flow ES was modeled using proxies of human usage, such as (1) a 30 km buffer zone to the nearest towns, (2) a 1 km buffer zone to vacation leases, (3) the area occupied by outfitters. The 30 km distance from towns was preferred over a distance decay function because in this remote region people have good knowledge of the land and tend to use specific spots for an ES repeatedly. These proxies, as well as those used to map the PUS, are context-specific and were weighted by previous social assessments and expert knowledge of human use of the territory (e.g. quantity of possible users and the permanency of use; (Hydro-Québec, 2007). For example, outfitters and vacation leases are strong predictors of demand for angling but are less predictive of wild fruit picking. Thus, a planning unit containing an outfitter and vacation leases will have a greater demand score for angling than a planning unit that does not contain these features.

2.3. Conservation assessment

2.3.1. Conservation planning software

Conservation networks were assembled using C-Plan v4.0 conservation planning software (Pressey et al., 2009). The C-Plan site selection algorithm is primarily based on irreplaceability measures, i.e. the likelihood that a given site will need to be selected in order to efficiently achieve conservation objectives (Kukkala and Moilanen, 2013). Planning units were selected first based on irreplaceability measures. When two or more sites had equal irreplaceability values, the area of the planning units was used as a proxy of cost (Naidoo et al., 2006) to identify the minimum set of sites that attain conservation targets for all features while minimizing the total selected area. One network was assembled per target level for each conservation scenario (see below).

2.3.2. Conservation scenarios

2.3.2.1. The biophysical supply scenario (BS). The biophysical supply scenario uses the BS maps of ES. Because the amount of the PUS for moose and duck hunting, salmon and trout angling, cloudberry picking and aesthetics is less than 27% of their BS supply (see Table 1), we decided to restrict the maximum targets of the BS scenario to 30%. Other targets tested were 5%, 10%, 15%, 20%, and 25% of their BS supply.

2.3.2.2. The potential-use supply scenario (PUS). The potential-use supply scenario uses the PUS maps of ES. Targets for moose and duck hunting, salmon and trout angling, cloudberry picking and aesthetics were set at 5%, 10%, 15%, 20%, 25%, 30%, 40%, 50%, 60% and 75% of their PUS supply. Because the PUS of cultural sites, flood control, carbon storage and the existence value of caribou ES had a greater spatial extent, we adjusted their targets in order to ensure that they did not disproportionally influence irreplaceability values or site selection (e.g. 99.4% of the study area is covered by planning units containing the PUS of carbon storage, while only 5.3% of the study area is covered by planning units containing the PUS of cloudberry picking). To proportionally weight their targets, we first calculated the proportion of the mean spatial extent of the PUS of the six local ES compared to the spatial extant of their PUS. These proportions were 0.25 for carbon storage, 0.35 for the existence value of caribou, 0.40 for flood control, and 0.33 for cultural sites. Finally, we multiplied each of the aforementioned conservation targets by these proportions to properly adjust the targets of these widespread ES.

2.3.2.3. The actual-use supply scenario using demand as a site selection rule (AUS-Algo). Demand for ES could be integrated into the site selection process by either assigning a conservation target to each ES demand or by including the total demand value for each planning unit as a rule in the site selection algorithms. Given that resources (land or money) available for conservation are limited worldwide (Balmford et al., 2003; Margules and Pressey, 2000), we used demand data for ES to assemble reserve networks that maximize demand fulfillment per unit of cost (i.e. demand-efficiency). Demand data for each local flow ES was standardized and summed for each planning unit. We did not consider demand for global and regional flow ES because their demand is equal across their PUS range; in other words, any selection of sites that achieves their supply targets will contribute equally to demand. The summed demand data was then integrated into the site selection algorithm in order to encourage the selection of sites with both high irreplaceability for ES supply and the highest summed demand. The maximization of demand rule was integrated before the minimization of area rule. Targets for all ES supply were the same as above.

2.3.2.4. The actual-use supply scenario using demand as a target (AUS-Target). In the AUS-Target scenario, conservation targets were set for the potential-use supply of all ES and specifically for local flow ES demand only. For the same reason as the AUS-Algo scenario, no demand targets were set for regional and global flow ES. Targets for ES supply were again set as 5%, 10%, 15%, 20%, 25%, 30%, 40%, 50%, 60% and 75% of their PUS. The same thresholds were used to set demand targets. Including targets for local flow ES demand should increase the irreplaceability value of sites containing both actual supply and demand of local flow ES, making them even more essential for meeting conservation objectives.

2.3.3. Conservation networks analysis

Conservation networks of all four scenarios were compared based on their performance in selecting the actual-use supply of ES and on their efficiency in securing ES demand (see Appendix B for network examples). Fostering the selection of the actual-use supply of ES does not necessarily guarantee that the conservation network will secure high demand value. Accordingly, efficiency is a complementary analysis in assessing whether or not our scenarios increase ES conservation effectiveness. We chose to focus on local flow ES since they are mainly affected by the spatial distribution of both their beneficiaries and demand. Therefore, even if the four conservation scenarios were assembled to secure the provision of the ten ES, we interpreted the results only for the six local flow ES, which are moose and duck hunting, salmon and trout angling, cloudberry picking and aesthetics. Considering regional and global flow ES could allow us to assess if gains in the effectiveness of local flow ES conservation could be achieved even when conservation planning does not exclusively focus on them. Moreover, we decided not to consider the cultural sites' ES with the other local flow ES in reporting the performance and efficiency results. This choice was made because this ES demand (and PUS) extends far beyond the PUS spatial range of the other local flow ES. Accordingly, almost any selected site would contribute to fulfilling demand for this ES. Therefore, this ES would have inflated the "true" performance and efficiency of conservation scenarios for securing local flow ES demand, particularly the BS and the PUS scenarios.

During a preliminary phase of this study, we also examined the performance of two other scenarios that targeted either ES demand or an index obtained by combining the standardized value of both ES supply and demand. However, because the demand value for a particular ES was not explicitly linked to the planning unit area, putting too much weight on ES demand in site selection resulted in networks containing a greater proportion of small sites with high demand values. Consequently, we chose not to report the results of these scenarios because they constantly secured an unpredictable low amount of ES supply.

Finally, no actual cost dataset was available for the study region. Using the planning unit area as proxy for cost may incorrectly assume that the costs are homogenous across the study area because areas with high demand (i.e. near where people live) are likely to be more expensive than other areas. However, while the use of actual cost data may have changed the spatial configuration of conservation networks, we believe that the interpretation of the results would not have diverged from the ones presented here. Hence, the minimization of area (or cost) rule in the C-Plan algorithm was applied to choose between sites having equal irreplaceability measures.

3. Results

3.1. Assessing the effects of mapping ES using a beneficiaries-based approach

The potential-use supply (PUS) of the six local flow ES (e.g. the supply currently accessible to humans) consisted of only 12%–27% of their total biophysical supply (BS; Table 1 and Fig. 2). In comparison, the PUS of regional flow ES (i.e. flood control) was 66% of its BS. Despite its local flow scale, the PUS of cultural sites for First Nations subsistence uptake was as much as 86% of its total BS. Given that the accessibility of sites is not an issue for global flow ES, the PUS and BS for both carbon storage and existence value were unsurprisingly similar. We compared the PUS scenario with the biophysical supply (BS) scenario to assess the effect of using a beneficiaries-based approach to map ES supply to prioritize areas for ES conservation. When considering only the PUS for selecting planning units, conservation networks contributed only partly to the actual-use supply of local flow ES (i.e. a combination of supply and demand of a particular ES). More precisely, only 45% of the selected planning units (Fig. 3(A)) or of their total area (Fig. 3(B)) contributed to secure an actual-use supply of local flow ES. Meanwhile, the BS scenario had only about 20% of its networks composed of planning units securing an actual-use supply of local flow ES. Going from targeting the biophysical supply to targeting the potential-use supply increased the chances of incidentally selecting a site with demand for local flow ES by decreasing the number of possible candidate sites considered at each iteration. Nevertheless, these results suggest that the BS scenario, and to a certain extent the PUS scenario, selected planning units that are not in demand or, more importantly, planning units that contain inaccessible local flow ES supply (see Fig. B1 in Appendix B). Nonetheless, even though demand was not used in both BS and PUS scenarios, the latter was two to three times more efficient than the former in choosing sites that fulfill demand for local flow ES (Fig. 4).

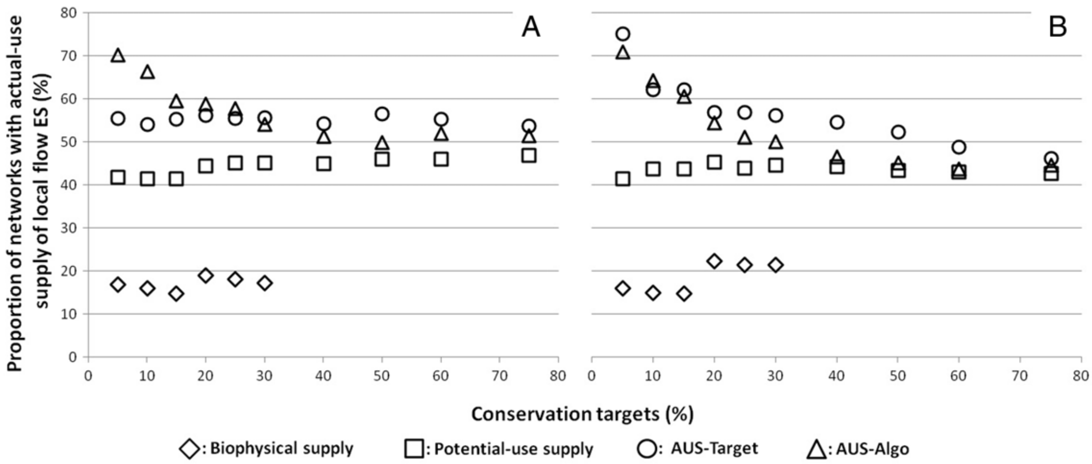

Fig. 3. Proportion of conservation network that secured the actual-use supply of local flow ES under the four conservation scenarios. One network was assembled at each target level for each scenario. The actual-use supply of an ES is the simultaneous presence of demand and access to supply in a planning unit. (A) The proportion per number of selected sites was calculated according to the number of planning units securing the actual-use supply of local flow ES divided by the total number of planning units in the network. (B) The proportion per total area selected was calculated according to the area contributing to secure the actual-use supply of local flow ES divided by the total area of the network. The biophysical supply scenario targeted only the biophysical supply of ES, the potential-use supply scenario targeted only the potential-use supply of ES (i.e. the accessible supply), the AUS-Target scenario targeted both ES potential-use supply and local flow ES demand, and the AUS-Algo targeted only ES potential-use supply among the sites with the highest summed demand. Networks were assembled targeting the ten ES (see Table 1) but the values are reported considering only the actual-use supply of the local flow ES (i.e. moose and duck hunting, salmon and trout angling, cloudberry picking and aesthetics).

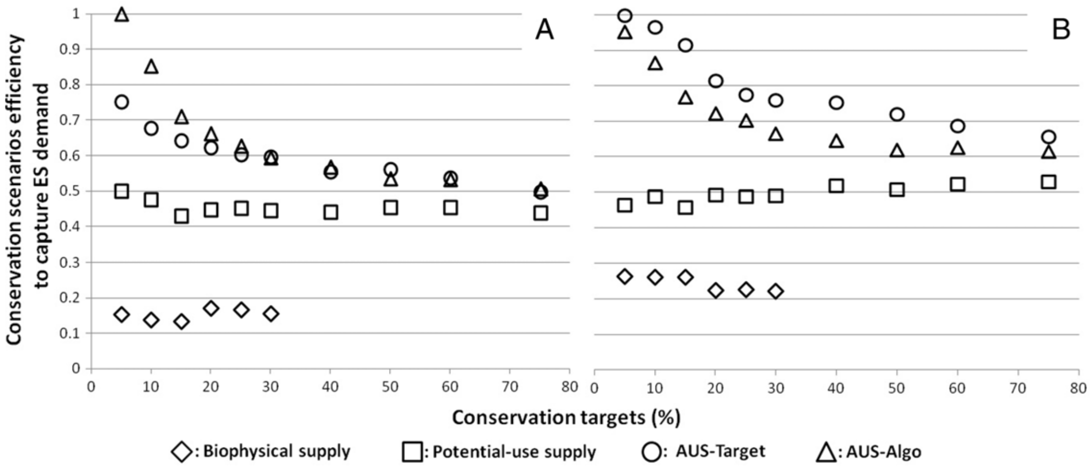

Fig. 4. Efficiency of four conservation scenarios to capture demand of local flow ES (see Fig. 3 for scenarios). (A) The efficiency of conservation scenarios per number of sites selected in the networks. (B) The efficiency of conservation scenarios per km^2 of selected sites in the networks. Efficiency values were calculated as the ratio of the total local flow ES demand secured per number of planning units selected in the networks or per km^2 of selected planning units in the networks. The efficiency values were standardized to enable easier comparisons of efficiency per number of selected sites with the efficiency per km^2 of selected sites. Networks were assembled targeting the ten ES (see Table 1) but the efficiency values are reported considering only local flow ES demand (i.e. moose and duck hunting, salmon and trout angling, cloudberry picking and aesthetics).

3.2. Integrating demand into identification of local flow ES priority areas

Scenarios considering demand, either as a selection rule (AUS-Algo) or as a target (AUS-Target), were compared with the PUS scenario to evaluate whether considering demand can further increase the effectiveness of ES conservation. At first glance, adding demand to conservation choices brought conservation areas closer to human populations (see Appendix B and Fig. 1). The advantages of using demand in identification of priority areas, rather than targeting potential-use supply (PUS) alone, were particularly apparent at low conservation targets (⩽30%; Fig. 3). As a result, consideration for demand forced the algorithm to restrict most of its choices to sites with demand for multiple local flow ES. However, at higher target levels the differences between the scenarios decreased as the chances of incidentally having a higher proportion of overlapping sites between them increased (see below). As a result, the AUS-Target and AUS-Algo scenarios established conservation networks that secured a higher proportion of actual-use supply of local flow ES (Fig. 3). Below targets of 30%, the efficiency of

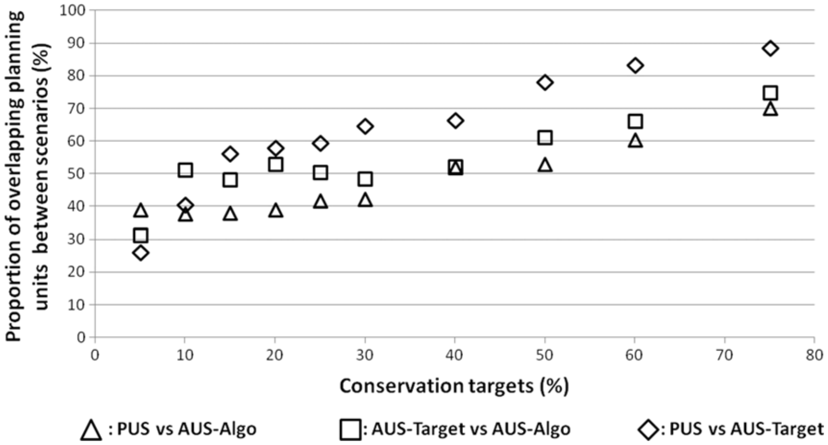

Fig. 5. Similarity between conservation scenarios. The proportion of overlapping planning units was calculated as a fraction of the number of selected sites in the smallest network considered in the comparison.

AUS-Target and the AUS-Algo scenarios to secure the actual-use supply of local flow ES was on average nearly 15% to 20% higher than the PUS scenario conservation networks (Fig. 3). Therefore, these two scenarios better ensured that the accessible supply of local flow ES would be useful to human beneficiaries (i.e. in demand) almost everywhere that supply is secured. Surprisingly, the PUS and AUS-Algo scenarios showed a similar proportion of actual-use supply secured per conservation network unit of area at conservation targets greater than 40% (Fig. 3(B)). This could be explained by the fact that at later stages of site selection under the AUS-Algo (and at high targets level), it was mostly the existence value of caribou and flood control targets that remained unachieved. At this stage, sites with the highest irreplaceability were mostly located outside the spatial range of local flow ES. Among these sites, the algorithm selected those that also had demand for cultural sites, while minimizing the total area selected. Nevertheless, cultural site demand was not considered in the calculation of the network' performance (Fig. 3). As a result, the networks' performance values per km^2 of AUS-Algo were slightly lowered.

Moreover, the planning units chosen with the two scenarios using demand typically secured a higher proportion of the total demand for local flow ES, as reflected by the higher efficiency of their networks in fulfilling demand (Fig. 4). As a direct result of our scenario's design, the AUS-Algo was more efficient in fulfilling demand when considering the number of planning units required to achieve targets (Fig. 4(A)), while the AUS-Target scenario was slightly more efficient per km^2 of selected planning units (Fig. 4(B)). Finally, integrating demand in the identification of priority areas led to a different composition and configuration of conservation networks, as suggested by the low proportion of planning units shared between scenarios (Fig. 5). More importantly, only 30% of the planning units selected by the two scenarios considering demand (AUS-Target and the AUS-Algo) overlapped spatially at a target of 10%. While similarity between conservation networks increased as conservation targets increased, only half of the sites overlapped for the two scenarios considering demand at targets ranging from 10% to 40%. This indicates that even the method chosen to integrate demand in identification of priority areas could introduce great spatial discrepancies in the resulting networks.

4. Discussion

4.1. Assessing the effects of mapping ES using a beneficiaries-based approach

By definition, ecosystem processes, structures and functions only give rise to ES where there are humans to benefit from them (Fisher et al., 2009; Potschin and Haines-Young, 2011). Therefore, to make effective and relevant conservation choices, it is important to spatially link ES supply to human beneficiaries. Our analyses showed that using a beneficiaries-based approach (the potential-use supply scenario) nearly doubled the proportion of sites providing an actual-use supply of local flow scale ES when compared to relying solely on the biophysical supply (the biophysical supply scenario). Mapping the biophysical supply (BS) could be useful for planning for future development and natural capital accounting. However, making immediate conservation choices using the BS would result in diluting and spending limited conservation funds on sites that do not provide benefits to humans. Using a beneficiaries-based approach to map the potential-use supply of each ES (PUS) should thus ensure that each site selected for ES protection provides accessible benefits.

In this study, proxies of human occupancy were used to assess which planning units provide an ES supply accessible to human populations. Mapping ES using this method showed that less than 27% of the total biophysical supply (i.e. PUS/BS ratio) of the six local flow ES was available for human use, reflecting a supply mostly inaccessible to humans in the study area. This is primarily due to the fact that the region is minimally developed and most human settlements are concentrated in the southern part (see Fig. 1). Future regional development for natural resources extraction is expected to occur in the study area (Berteaux, 2013). As development occurs, the resulting expansion of road networks will improve accessibility to the territory. Accordingly, as new ES supply becomes available to human populations, the PUS/BS ratio of local flow ES will increase and new conservation opportunities will be created. However, despite the increase in the PUS supply of local flow ES, land cover changes and ecosystem conversions resulting from development will also cause a net decrease in the study area's BS of all ES. The extent of anthropogenic disturbances can indeed have a huge impact on the capacity of ecosystems to provide different categories of ES (Cimon-Morin et al., 2013; de Groot et al., 2010; Foley et al., 2005). For instance, regulating (e.g. carbon storage and flood control) and some cultural ES (e.g. the existence value of caribou) are considered to be at a maximum in natural or slightly used ecosystems, while the flow of provisioning ES are considered to be non-existent to low in such ecosystems.

4.2. Integrating demand into identification of local flow ES priority areas

For any given conservation target, there are often multiple combinations of sites (solutions) that would make it possible to achieve these objectives. This is especially true for regions where many undisturbed or natural sites are still available for conservation, as in our study area. Choosing the best solution among all possible alternatives may require the inclusion of new criteria directly into the systematic conservation planning procedure. In this study, we posited that using demand for ES could increase the effectiveness of the priority areas identified by fostering the selection of sites where there is a high level of need for these supplies. Using demand caused great spatial discrepancies, compared to the conservation networks assembled using only the potential-use supply scenario (PUS), particularly at low representation targets (Fig. 5). This supports the suggestion that there is a spatial trade-off between ES supply and demand (Burkhard et al., 2012; Chan et al., 2006; Holland et al., 2011) and clearly illustrates its consequences for conservation assessments. A very different set of selected sites could result from approaches that include demand as opposed to those that do not. Furthermore, this indicates that mapping ES using a beneficiaries-based approach may not be sufficient for efficient conservation of local flow ES demand particularly, since actual conservation targets are often low due to the lack of available resources.

In order to assess how to best use demand data, we tested two different approaches: (1) the AUS-Algo, for which conservation networks were assembled to maximize demand fulfillment (2) the AUS-Target, where ES demand was used as targeted features. The comparison of overlapping sites selected by both scenarios also showed great spatial discrepancies (Fig. 5). This illustrates that the method used to include demand can result in very different conservation networks. The AUS-Algo does not guarantee that each particular ES demand will be sufficiently or equally represented, but rather ensures that each new planning unit added to the network will be selected among the ones with the highest summed demand of local flow ES, regardless of which ES demand is currently fulfilled. Thus, this approach may not be suitable when the spatial congruence of different ES demands is low. On the other hand, using demand as targets conserved at least a minimal amount of demand for each ES, but its resulting networks were larger in both total area and number of planning units selected than when integrating demand into the selection process. Targets for demand were perhaps set too high in comparison to ES supply's targets, and additional area was needed to achieve the local flow ES demand targets. However, in the context of our study, setting targets for ES demand seemed to be the best way of ensuring that each ES demand had a specific degree of representation (i.e. an exact amount) secured into conservation networks.

Conservation actions often create significant economic loss in the form of opportunity costs to local human populations by causing the foreclosure of future land-use options (Adams et al., 2004; Linnell et al., 2011). Thus, when projected conservation costs outweigh benefits, there is a risk that conservation will be limited to distant, unproductive and uninteresting localities (Moilanen et al., 2009). Nevertheless, ES provide a means for valuing human's well-being in conservation projects and can contribute to improve the societal acceptance and implementation of conservation actions (Cimon-Morin et al., 2014; Goldman et al., 2008) by making local communities benefit from them. ES offer the possibility to better align conservation and human usage of ecosystems by enabling the pursuit of some local population livelihood activities linked to nature in protected areas. Provisioning services conservation tends more towards the maintenance and sustainable uptake of harvestable species rather than the preservation of biodiversity specifically. For example, setting adequate conservation targets for moose supply should ensure the conservation of its habitats (or populations), while the use of demand will protect moose populations where they can also be hunted by beneficiaries. Thereby, our results showed that using demand in site selection procedures could further increase the number of local people who benefit from conservation when compared to the PUS scenario. It resulted indeed in conservation networks containing a higher proportion of planning units providing potential-use supply and having a higher efficiency in fulfilling demand of local flow ES (Figs. 3 and 4). Thus, the effective conservation of local flow ES could justify the need to bring conservation actions closer to human population, that is to say, in the most threatened and costly ecosystem to protect.

Similar to ES supply, demand for ES is not static and is likely to change over time as the region's population grows larger and expands spatially with the spread of human settlements and infrastructures (e.g. more vacation leases, residential development, expansion of road networks, etc.). Nevertheless, considering the dynamic nature of both ES supply and demand, we believe that regional development should not proceed at the expense of ES conservation. Since demand for local flow ES decreases with increasing distance from beneficiaries (Chan et al., 2006; Holland et al., 2011), the protection of more distant sites (or newly accessible distant sites) may not adequately fulfill demand and is not likely to be sufficient to sustain current and future levels of well-being for most local ES users.

In this study, spatially-defined field data about demand (i.e. primary data) was available for only two ES (moose hunting and salmon angling), while secondary data was used to estimate demand for the other local flow ES. Such use of secondary data resulted in maps with a higher proportion of planning units sharing the same demand value, e.g. similar in terms of composition in the proxies used for estimating demand. The use of secondary data to map ES demand thus creates uncertainty in reserve selection because it does not ensure that the planning units that meet the criteria for high demand are actually used more by humans than others. For example, the comparison of primary demand data for moose hunting with demand scores that could have been obtained using our secondary data modeling method showed a moderate positive correlation ($r = 0.32$, $p < 0.0001$; results are not shown). Likewise, it has also been reported that using secondary data to map ES supply could hinder the identification of priority areas (Eigenbrod et al., 2010a,b). Although primary data should enable more relevant conservation choices, our approach illustrates the imperative of considering demand in systematic conservation planning of ES.

5. Conclusion

In order to halt the global loss of ES supply, efforts have been made to include ES in conservation assessments (Chan et al., 2006; Egoh et al., 2007). However, shifting the focus of conservation to safeguarding human well-being also requires broadening traditional conservation goals to better spatially link conservation actions to human beneficiaries. In this study, we posited that ES conservation networks should secure an accessible ES supply, in a location where ES are greatly needed by their human beneficiaries. We showed that targeting the actual-use supply of ES, which is a combination of ES potential-use supply and demand, using systematic conservation planning procedures enabled conservation networks to achieve ES conservation objectives more effectively. For the time being, setting conservation targets for ES supply and demand seems the best approach for selecting the actual-use supply of ES, particularly in regard to other conservation goals such as cost-efficiency. Our results constitute a first step towards adapting SCP procedures for ES conservation.

Despite the fact that this study was conducted in a remote region, our results are relevant for ES conservation assessments worldwide for a number of reasons. In more human dominated regions, for example, local flow ES may provide accessible benefits to humans almost everywhere they are supplied, yet their demand remains spatially heterogeneous. Therefore, as we demonstrated, failure to consider demand during site selection could result in choosing planning units that are not the most efficient with respect to demand fulfillment, and ultimately achieving ES conservation objectives. Moreover, priority areas for biodiversity and ES conservation tend to lack spatial congruence globally (Cimon-Morin et al., 2013). Since funds available for conservation are often limited, even a slight increase in the effectiveness of ES conservation is critical to lowering the impact on resource allocation and better aligning biodiversity and ES conservation.

Acknowledgments

We would also like to thank our major project partners for their financial and technical support: Ducks Unlimited Canada; Laval University; the Ministry of Sustainable Development, Environment, Wildlife and Parks (MDDEFP); Hydro-Québec (contract reference 37596-10021); Quebec Research Funds for Nature and Technology (FRQNT); and the Natural Sciences and Engineering Research Council of Canada (NSERC; BMP grant to JCM [demand No 163396] and Discovery grant No 312195 to MP). We also thank S. Boudreau for her comments on the manuscript. Finally, we wish to thank Karen Grislis for the stylistic revision of this manuscript and two anonymous reviewers.

Appendix A. Supplementary data

Supplementary material related to this article can be found online at http://dx.doi.org/10.1016/j.gecco.2014.07.005.

References

Adams, W.M., Aveling, R., Brockington, D., Dickson, B., Elliott, J., Hutton, J., Roe, D., Vira, B., Wolmer, W., 2004. Biodiversity conservation and the eradication of poverty. Science 306, 1146–1149.

Bagstad, K.J., Johnson, G.W., Voigt, B., Villa, F., 2013. Spatial dynamics of ecosystem service flows: A comprehensive approach to quantifying actual services. Ecosyst. Serv. 4, 117–125.

Balmford, A., Fisher, B., Green, R.E., Naidoo, R., Strassbrug, B., Turner, R.K., Rodrigues, A.S.L., 2011. Bringing ecosystem services into the real world: An operational framework for assessing the economic consequences of losing wild nature. Environ. Resour. Econ. 48, 161–175.

Balmford, A., Gaston, K.J., Blyth, S., James, A., Kapos, V., 2003. Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. Proc. Natl. Acad. Sci. 100, 1046–1050.

Balvanera, P., Pfisterer, A.B., Buchmann, N., He, J.-S., Nakashizuka, T., Raffaelli, D., Schmid, B., 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecol. Lett. 9, 1146–1156.

Bellavance, M.F., Gagné, J., 2012. CRRNT Côte-Nord: carte des lacs sans Poisson de la Minganie. In: Conférence régionale des élus de la Côte-Nord, Baie-Comeau.

Berteaux, D., 2013. Québec's large-scale plan nord. Conserv. Biol. 27, 242–247.

Burkhard, B., Kroll, F., Nedkov, S., Müller, F., 2012. Mapping ecosystem service supply, demand and budgets. Ecol. Indicators 21, 17–29.

Caron, F., Fontaine, P.-M., Cauchon, V., 2006. État des stocks de saumon au Québec en 2005. Ministère des Ressources naturelles et de la Faune du Québec. In: Direction de la recherche sur la faune. p. 18.

Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C., Daily, G.C., 2006. Conservation planning for ecosystem services. PLoS Biol. 4, 2138–2152.

Chapin, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C., Diaz, S., 2000. Consequences of changing biodiversity. Nature 405, 234–242.

Charest, P., 1996. Les stratégies de chasse des Mamit Inuat. Anthropol. Soc. 20, 107–128.

Charest, P., 2005. L'organisation socio-territoriale de la chasse. In: Charest, P., Huot, J., McNulty, G. (Eds.), Les Montagnais et la faune. Université Laval, Sainte-Foy, Québec, pp. 235–353. Chapitre 4.

Cimon-Morin, J., Darveau, M., Poulin, M., 2013. Fostering synergies between ecosystem services and biodiversity in conservation planning: A review. Biol. Cons. 166, 144–154.

Cimon-Morin, J., Darveau, M., Poulin, M., 2014. Ecosystem services expand the biodiversity conservation toolbox - A response to Deliège and Neuteleers. Biol. Cons. 172, 219–220.

de Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol. Complexity 7, 260–272.

Ducruc, J.-P., 1985. L'analyse écologique du territoire au Québec: L'inventaire du capital-nature de la moyenne-et-basse-côte-nord. Environnement Québec - Environnement Canada - Hydro-Québec, Division des inventaires écologiques, Série de l'inventaire du capital-nature. Numéro 6.

Dudley, N., 2008. Guidelines for Applying Protected Area Management Categories. IUCN, Gland, Switzerland, pp. x + 86.

Dyer, S.J., O'Neill, J.P., Wasel, S.M., Boutin, S., 2001. Avoidance of industrial development by Woodland Caribou. J. Wildl. Manage. 65, 531–542.

Egoh, B., Reyers, B., Rouget, M., Richardson, D.M., Le Maitre, D.C., van Jaarsveld, A.S., 2008. Mapping ecosystem services for planning and management. Agric. Ecosyst. Environ. 127, 135–140.

Egoh, B., Rouget, M., Reyers, B., Knight, A.T., Cowling, R.M., van Jaarsveld, A.S., Welz, A., 2007. Integrating ecosystem services into conservation assessments: A review. Ecol. Econom. 63, 714–721.

Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Heinemeyer, A., Gillings, S., Roy, D.B., Thomas, C.D., Gaston, K.J., 2010a. Error propagation associated with benefits transfer-based mapping of ecosytem services. Biol. Cons. 143, 2487–2493.

Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Heinemeyer, A., Gillings, S., Roy, D.B., Thomas, C.D., Gaston, K.J., 2010b. The Impact of proxy-based methods on mapping the distribution of ecosystem services. J. Appl. Ecol. 47, 377–385.

Environment Canada, 2008. Scientific review for the identification of critical habitat for woodland caribou (Rangifer tarandus caribou), B. P., in Canada. Environment Canada, Ottawa, Canada. 72 pp. plus 180 pp Appendices.

Environmental Systems Research Institute Inc. (ESRI), 2012. ArcGIS 10.0 Redlands, USA.

Ferland, M.E., del Giorgio, P.A., Teodoru, C.R., Prairie, Y.T., 2012. Long-term C accumulation and total C stocks in boreal lakes in northern Quebec. Glob. Biogeochem. Cycles 26.

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecol. Econom. 68, 643–653.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science 309, 570–574.

Fortin, D., Buono, P.L., Fortin, A., Courbin, N., Gingras, C.T., Moorcroft, P.R., Courtois, R., Dussault, C., 2013. Movement responses of caribou to human-induced habitat edges lead to their aggregation near anthropogenic features. Amer. Natur. 181, 827–836.

Goldman, R.L., Tallis, H., Kareiva, P., Daily, G.C., 2008. Field evidence that ecosystem service projects support biodiversity and diversify options. Proc. Natl. Acad. Sci. 105, 9445–9448.

Gouvernement du Québec, 1993. Hydrologie, In Manuel de conception de ponceaux. Gouvernement du Québec, Ministère des Transports, direction des structures, service de l'hydraulique.

Gouvernement du Québec, 2013. Région 09: Côte-Nord. MRC et agglomérations ou municipalités locales exerçant certaines compétences de MRC. Ministère des affaire municipales, régions et occupation du territoire. Direction de la géomatique et de la statistique. http://www.mamrot.gouv.qc.ca/pub/organisation_municipale/cartotheque/Region_09.pdf.

Guérette Montminy, A., Berthiaume, E., Darveau, M., Cumming, S., Bordage, D., Lapointe, S., Lemelin, L.V., 2009. Répartition de la sauvagine en période de nidification entre les 51^0 et 58^0 de latitude nord dans la province de Québec, Canards Illimités Canada - Québec, Rapport technique no Q14, Québec, p. 43.

Guo, Z., Zhang, L., Li, Y., 2010. Increased dependence of humans on ecosystem services and biodiversity. Plos One 5.

Holland, R.A., Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Thomas, C.D., Heinemeyer, A., Gillings, S., Roy, D.B., Gaston, K.J., 2011. Spatial covariation between freshwater and terrestrial ecosystem services. Ecol. Appl. 21, 2034–2048.

Horwath, W.R., 2007. Carbon cycling and formation of soil organic matter. In: Paul, E.A. (Ed.), Soil microbiology, Ecology And Biochemistry. Academic Press, New York, pp. 303–390.

Hydro-Québec, 2007. Complexe de la Romaine - Étude d'impact sur l'environnement. Hydro-Québec Équipement et Hydro-Québec Production, http://www.hydroquebec.com/romaine/documents/etude.html.

Kukkala, A.S., Moilanen, A., 2013. Core concepts of spatial prioritisation in systematic conservation planning. Biol. Rev. 88, 443–464.

Lamontagne, G., Lefort, S., 2004. Plan de gestion de l'orignal 2004–2010, Ministère des Ressources naturelles, de la Faune et des Parcs du Québec, Québec, p. 265.

Larsen, F.W., Londoño-Murcia, M.C., Turner, W.R., 2011. Global priorities for conservation of threatened species, carbon storage, and freshwater services: scope for synergy? Conserv. Lett. 4, 355–363.

Lemelin, L.V., Bordage, D., Darveau, M., Lepage, C., 2004. Répartition de la sauvagine et d'autres oiseaux utilisant les milieux aquatiques en période de nidification dans le Québec forestier, Environnement Canada, Service canadien de la faune, Série de rapports techniques n^o422, Sainte-Foy, p. 70.

Lemelin, L.V., Darveau, M., 2008. Les milieux humides du parc national du Canada de la Mauricie: cartographie en vue d'une surveillance de l'intégrité écologique. Ducks Unlimited Canada, Québec.

Lemelin, L.V., Darveau, M., Imbeau, L., Bordage, D., 2010. Wetland use and selection by breeding waterbirds in the boreal forest of Quebec, Canada. Wetlands 30, 321–332.

Li, T., Ducruc, J.-P., 1999. Les provinces naturelles. Niveau I du cadre écologique de référence du Québec. Ministère de l'Environnement, 90 p. http://www.mddefp.gouv.qc.ca/biodiversite/aires_protegees/provinces/index.htm.

Linnell, J.D.C., Rondeau, D., Reed, D.H., Williams, R., Altwegg, R., Raxworthy, C.J., Austin, J.D., Hanley, N., Fritz, H., Evans, D.M., Gordon, I.J., Reyers, B., Redpath, S., Pettorelli, N., 2011. Confronting the costs and conflicts associated with biodiversity. Anim. Conserv. 13, 429–431.

Luck, G.W., Chan, K.M.A., Fay, J.P., 2009. Protecting ecosystem services and biodiversity in the world's watersheds. Conserv. Lett. 2, 179–188.

Maes, J., Paracchini, M.L., Zulian, G., Dunbar, M.B., Alkemade, R., 2012. Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. Biol. Cons. 155, 1–12.

Magnan, G., Garneau, M., Payette, S., 2011. Paleohydrologyand long-term carbon dynamics in the ombrotrophic peatlandsof the North Shore of the Saint-Lawrence, Northeastern Québec, In: International Symposium on Responsible Peatland Management and Growing Media Production. Québec, Canada.

Margules, C.R., Pressey, R.L., 2000. Systematic conservation planning. Nature 405, 243–253.

Margules, C.R., Sarkar, S., 2007. Systematic Conservation Planning. Cambridge University Press, Cambridge.

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

Moilanen, A., Arponen, A., Stokland, J.N., Cabeza, M., 2009. Assessing replacement cost of conservation areas: How does habitat loss influence priorities? Biol. Cons. 142, 575–585.

MRNF, 2012a. Bilan de l'exploitation du saumon au Québec 2011. Ministère des Ressources naturelles et de la Faune, Secteur Faune Québec, Secteur des opérations régionales.

MRNF, 2012b. La pêche sportive au Québec – Périodes, limites et exceptions - Zone 19 sud, p. 14 p. Ministère des Ressources Naturelles et de la Faune.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R., Ricketts, T.H., 2008. Global mapping of ecosystem services and conservation priorities. Proc. Natl. Acad. Sci. 105, 9495–9500.

Naidoo, R., Balmford, A., Ferraro, P.J., Polasky, S., Ricketts, T.H., Rouget, M., 2006. Integrating economic costs into conservation planning. Trends Ecol. Evol. 21, 681–687.

Natural Resources Canada (NRC) 2011. CanVec v8.0. Natural Resources Canada, Earth Science Sector, Centre for Topographic Information. Available online via Géogratis: http://geogratis.cgdi.gc.ca/geogratis/fr/collection/5460AA9D-54CD-8349-C95E-1A4D03172FDF.html.
Nedkov, S., Burkhard, B., 2012. Flood regulating ecosystem services—Mapping supply and demand, in the Etropole municipality, Bulgaria. Ecol. Indicators 21, 67–79.
Pâquet, J., 1997. Analyses du paysage visuel, In Application de la cartographie écologique à quelques éléments de la gestion forestière, In: Bissonnette, J., Gerardin, V., Pâquet, J. (Eds.), Ministère de l'Environnement et de la Faune du Québec, Direction de la conservation et du patrimoine écologique, Québec, pp. 41–74.
Pâquet, J., 2003. Outil d'aide à la décision pour classifier les secteurs d'intérêt majeurs et définir les stratégies d'aménagement pour l'intégration visuelle des coupes dans les paysages – Objectif de protection ou de mise en valeur des ressources du milieu forestier visant le maintien de la qualité visuelle des paysages forestiers. Ministère des Ressources Naturelles, de la Faune et des Parcs du Québec, Direction des programmes forestiers, p 15.
Pâquet, J., Bélanger, L., 1998. Stratégie d'aménagement pour l'intégration visuelle des coupes dans les paysages. Réalisé par C.A.P. Naturels dans le cadre du Programme de mise en valeur des ressources du milieu forestier du ministère des Ressources naturelles. Charlesbourg, p. 40.
Potschin, M., Haines-Young, R., 2011. Ecosystem services: Exploring a geographical perspective. Prog. Phys. Geogr. 35, 575–594.
Pressey, R.L., Bottrill, M.C., 2008. Opportunism, threats, and the evolution of systematic conservation planning. Conserv. Biol. 22, 1340–1345.
Pressey, R.L., Watts, M.E., Barrett, T.W., Ridges, M.J., 2009. The C-Plan conservation planning system: origins, applications, and possible futures. In: Moilanen, A., Wilson, K.A., Possingham, H.P. (Eds.), Spatial Conservation Prioritization. Oxford University Press, Oxford, pp. 211–234.
Reyers, B., Polasky, S., Tallis, H., Mooney, H.A., Larigauderie, A., 2012. Finding common ground for biodiversity and ecosystem services. Bioscience 62, 503–507.
Sarkar, S., Illoldi-Rangel, P., 2010. Systematic conservation planning: an updated protocol. Natur. Conserv. 8, 19–26.
Saucier, J.-P., Grondin, P., Robitaille, A., Gosselin, J., Morneau, C., Richard, P.J.H., Brisson, J., Sirois, L., Leduc, A., Morin, H., Thiffault, É., Gauthier, S., Lavoie, C., Payette, S.,. Écologie forestière, In Manuel de foresterie, 2[e] éd. Ouvrage collectif. ed. Ordre des ingénieurs forestiers du Québec, Éditions MultiMondes, Québec, pp. 165–316.
Seip, D.R., Johnson, C.J., Watts, G.S., 2007. Displacement of mountain caribou from winter habitat by snowmobiles. J. Wildl. Manage. 71, 1539–1544.
Tallis, H., Polasky, S., 2009. Mapping and valuing ecosystem services as an approach for conservation and natural-resource management. Ann. New York Acad. Sci. 1162, 265–283.
Tallis, H., Polasky, S., Lozano, J.S., Wolny, S., 2012. Inclusive wealth accounting for regulating ecosystem services. In: UNU-IHDP and UNEP. Inclusive Wealth Report 2012. Measuring Progress towards Sustainability. Cambridge University Press, Cambridge.
Tarnocai, C., Lacelle, B., 1996. Soil Organic Carbon Digital Database of Canada. Eastern Cereal and Oilseed Research Center, Research Branch, Agriculture and Agri-Food Canada, Ottawa, Canada.
Tecsult, 2006. Raccordement du complexe de la Romaine – Étude des populations de caribous et d'orignaux. Rapport final présenté à Hydro-Québec Équipement. Pagination multiple + annexes.
Timmermann, H.R., McNicol, J.G., 1988. Moose Habitat Needs. Forestry Chronicle 64, 238–245.
Turner, W.R., Brandon, K., Brooks, T.M., Costanza, R., da Fonseca, G.A.B., Portela, R., 2007. Global conservation of biodiversity and ecosystem services. Bioscience 57, 868–873.
van Jaarsveld, A.S., Biggs, R., Scholes, R.J., Bohensky, E., Reyers, B., Lynam, T., Musvoto, C., Fabricius, C., 2005. Measuring conditions and trends in ecosystem services at multiple scales: the Southern African Millennium Ecosystem Assessment (SAfMA) experience. Philos. Trans. R. Soc. B 360, 425–441.
Vistnes, I., Nellemann, C., 2007. Impacts of human activity on reindeer and caribou: The matter of spatial and temporal scales. In: Rangifer Rep. pp. 47–56.
Vistnes, I., Nellemann, C., 2008. The matter of spatial and temporal scales: a review of reindeer and caribou response to human activity. Polar Biol. 31, 399–407.
Vitousek, P.M., Mooney, H.A., Lubchenco, J., Melillo, J.M., 1997. Human domination of Earth's ecosystem. Science 277, 494–499.
Vors, L.S., Schaefer, J.A., Pond, B.A., Rodgers, A.R., Patterson, B.R., 2007. Woodland caribou extirpation and anthropogenic landscape disturbance in Ontario. J. Wildl. Manage. 71, 1249–1256.
Walsh, G., 2005. In: Charest, P., Huot, J., McNulty, G. (Eds.), La récolte faunique, In Les Montagnais et la faune. Université Laval, Sainte-Foy, Québec, pp. 355–422. Chapitre 5.