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Accounting for capacity and flow of ecosystem services: A conceptual model and a case study for Telemark, Norway

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ABSTRACT

Understanding the flow of ecosystem services and the capacity of ecosystems to generate these services is an essential element for understanding the sustainability of ecosystem use as well as developing ecosystem accounts. We conduct spatially explicit analyses of nine ecosystem services in Telemark County, Southern Norway. The ecosystem services included are moose hunting, sheep grazing, timber harvest, forest carbon sequestration and storage, snow slide prevention, recreational residential amenity, recreational hiking and existence of areas without technical interference. We conceptually distinguish capacity to provide ecosystem services from the actual flow of services, and empirically assess both. This is done by means of different spatial models, developed with various available datasets and methods, including (multiple layer) look-up tables, causal relations between datasets (including satellite images), environmental regression and indicators derived from direct measurements. Capacity and flow differ both in spatial extent and in quantities. We discuss five conditions for a meaningful spatial capacity–flow-balance. These are (1) a conceptual difference between capacity and flow, (2) spatial explicitness of capacity and flow, (3) the same spatial extent of both, (4) rivalry or congestion, and (5) measurement with aligned indicators. We exemplify spatially explicit balances between capacity and flow for two services, which meet these five conditions. Research in the emerging field of mapping ES should focus on the development of compatible indicators for capacity and flow. The distinction of capacity and flow of ecosystem services provides a parsimonious estimation of over- or underuse of the respective service. Assessment of capacity and flow in a spatially explicit way can thus support monitoring sustainability of ecosystem use, which is an essential element of ecosystem accounting.

1. Introduction

1.1. Background

The concept of ecosystem services (ES) is increasingly used to analyse the human–nature relationship and inform policy makers and land-use planners in order to support sustainable use of ecosystems (Carpenter et al., 2009; Daily et al., 2009; De Groot et al., 2010; Larigauderie et al., 2012). Among different policy instruments that can be supported by the ES concept, ecosystem accounting, with the aim of monitoring extent, condition and properties of ecosystems that deliver ES over time in both monetised and non-monetised values, has recently drawn increased attention (Boyd and Banzhaf, 2007; Edens and Hein, 2013; EEA, 2010; Jordan et al., 2010; Mäler et al., 2008; Stoneham et al., 2012; ten Brinck, 2011; Weber, 2007). The recent System of Environmental-Economic Accounting Experimental Ecosystem Accounting (SEEA) guidelines define ecosystem accounting as "an approach to the assessment of the environment through the measurement of ecosystems, and measurement of the flows of services from ecosystems into economic and other human activity" (European Commission, 2013). Several challenges still remain to be addressed regarding standardising methodology for biophysical ecosystem accounting (Boyd and Banzhaf, 2007; European Commission, 2013; Stoneham et al., 2012). Among these are (i) clarity of concepts in order to monitor ES in a scientifically correct and practically feasible manner, (ii) accuracy and use of representative indicators at large spatial scales in face of data limitations, and (iii) the spatial explicitness of ES.

1.1.1. Conceptual clarity in the distinction of capacity and flow

Conceptual clarity, measurability and robustness of terms and definitions are demanded for accounting systems that need to

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monitor and measure ES over longer periods of time. Recent conceptualisations of ES have highlighted the need for distinguishing the capacity to provide services and their actual use (Burkhard et al., 2012; De Groot et al., 2010; Haines-Young and Potschin, 2010a; van Oudenhoven et al., 2012). This distinction between capacity and flow of ES has the potential to deliver a practical, policy-relevant measure of sustainability, but remains to be clarified in terms of definitions and tested empirically (Schröter et al., 2012).

1.1.2. Scale, accuracy and indicators for ecosystem accounting

Larger spatial scales of studies are especially interesting for policy instrument development, general frameworks for land-use policy and monitoring and accounting for ES, as these usually are applied to larger institutional units (counties, provinces, states). Furthermore, a higher spatial scale allows for including many different ecosystems (Turner et al., 1989) and beneficiaries who often live far from ecosystems that deliver services (Borgström Hansson and Wackernagel, 1999). However, spatially representative data at high resolutions is less likely to be found across larger areas. As a consequence the resolution of ES maps at higher spatial scales found in the literature is often low, and the employed models allow for little consideration of spatial variability. As a result of low data availability at higher spatial scales either qualitative instead of quantitative methods have been applied (e.g. Burkhard et al., 2012; Haines-Young et al., 2012) or ES proxies (Eigenbrod et al., 2010) and indicators with low ability to convey information were chosen (Layke et al., 2012). However, indicators that are able to represent indicated object and progress towards policy goals (Kandziora et al., 2013; Müller and Burkhard, 2012), cover relevant cause–effect relations, and are accurate and reliable are highly needed for the development of policy instruments like ecosystem accounting (Edens and Hein, 2013; Gómez-Baggethun and Barton, 2013).

1.1.3. Spatially explicit assessments of multiple ES

Spatial explicitness of both provision by ecosystems and actual use of ES by society is a crucial characteristic of ES (Costanza, 2008; Fisher et al., 2009; Hein et al., 2006; Schröter et al., 2012). Accordingly, a spatial approach to ES can contribute to the development of decision support tools with ecosystem accounting as a case in point. Spatial restrictions such as accessibility, remoteness or proximity of ecosystems also determine the state, use and value of ES (Balmford et al., 2008; Bateman, 2009; Boyd, 2008; Fisher et al., 2009; Troy and Wilson, 2006). Such restrictions have rarely been demonstrated empirically. Mapping of multiple ES has become an important scientific endeavour, while the number of ES considered in studies still remains low and validation of results is rarely carried out (Seppelt et al., 2011). While the importance of cultural ES has frequently been pointed out, many of these services have yet to be adequately defined, quantified and made compatible with a larger set of ES (Chan et al., 2012; Daniel et al., 2012).

1.2. Article aims

The objective of this study is to test and validate spatial capacity and flow models of multiple ES for ecosystem accounting purposes. We conceptually distinguish capacity and flow and introduce this distinction as a parsimonious measure for sustainability. Indicator choice is critical for the analysis of ES. For the purpose of analysing sustainability of the capacity–flow relation of ES we therefore develop, test and discuss suitable indicators. Our empirical quantification approach is tested on a provincial scale for Telemark County in southern Norway. While there is growing interest in applying the ES concept in different regions of the world, there is little knowledge on ES from hemi-boreal, mountainous countries such as Norway (Barton et al., 2011). The institutional scale of a county seems appropriate, as it is large enough to test large-scale spatial ES models, including many different ecosystem types. The temporal scale of our study is one year (2010). We thereby do not consider variations of ES capacity and flow within a year or across years.

2. Methodology and materials

2.1. Defining spatial ecosystem accounting

The main aim of ecosystem accounting is to monitor changes in ecosystem conditions and ES over time from a spatial perspective in a way that is consistent with national accounting (Fig. 1, and European Commission, 2013). Furthermore, accounting for socio-economic contributions to the existence of ES is partly, but not systematically, done in conventional accounting, e.g. in the case of harvesting machines, or tourist overnight stays. The left part of Fig. 1 (measurement of ecosystems) comprises spatial extent and characteristics or properties of ecosystems, which are included as quantitative and spatial model inputs in this study. The focus of this study is the spatial quantification of ES during one year, making use of both ecosystem and socio-economic data (Fig. 1). Spatially explicit accounting needs to be structured in geographic units. In accordance with the SEEA guidelines, we define the County of Telemark as the ecosystem accounting unit. It is divided into land cover/ecosystem functional units for which we take a satellite-derived map comprising 25 vegetation types. This land cover data set is based on classified Landsat 5/TM and Landsat 7/ETM+ satellite images and was created by integrating topographical information and a standardised vegetation mapping system (Johansen, 2009). The land cover units are sub-divided into basic spatial units (100 m by 100 m grains) for which a service-load per unit can be determined for each ES. This resolution was chosen to reflect an appropriate level of spatial variability while at the same time being able to handle big data volumes.

2.2. Distinguishing ES capacity and flow

Following Haines-Young and Potschin (2010b), ES are "the contributions that ecosystems make to human well-being, and arise from the interaction of biotic and abiotic processes." The term contributions refers to the fact that the final use of many ES can only take place after economic agents (e.g. ecosystem managers, primary resource exploiters, private persons) have modified ecosystems or actually harvested services. It is possible to determine a point in time and space of the last contribution of the ecosystem. Contributions are those properties of an ecosystem that are appreciated by humans (e.g. certain population sizes, regrowth rates, certain ecosystem states) and which are based on the results of different transfers of energy, matter and information (ecosystem processes). Because of restrictions such as low spatial accessibility, absence of beneficiaries or low management pressure, not all ecosystem properties constitute an ES. Furthermore, actual use of ES can exceed the flows that ecosystems can potentially generate within a certain time period so that for instance stocks are depleted. We therefore distinguish two aspects in the emergence of an ES: ES capacity and ES flow. ES capacity is the long-term potential of ecosystems to provide services appreciated by humans in a sustainable way, under the current management of the ecosystem. Many ecosystems are in fact social–ecological systems (Ostrom, 2009) as modifications (of the potential to provide ES) by humans are already present. Capacity may be increased or decreased over time through ecosystem management and land use conversion, but we do not focus here on different management options.

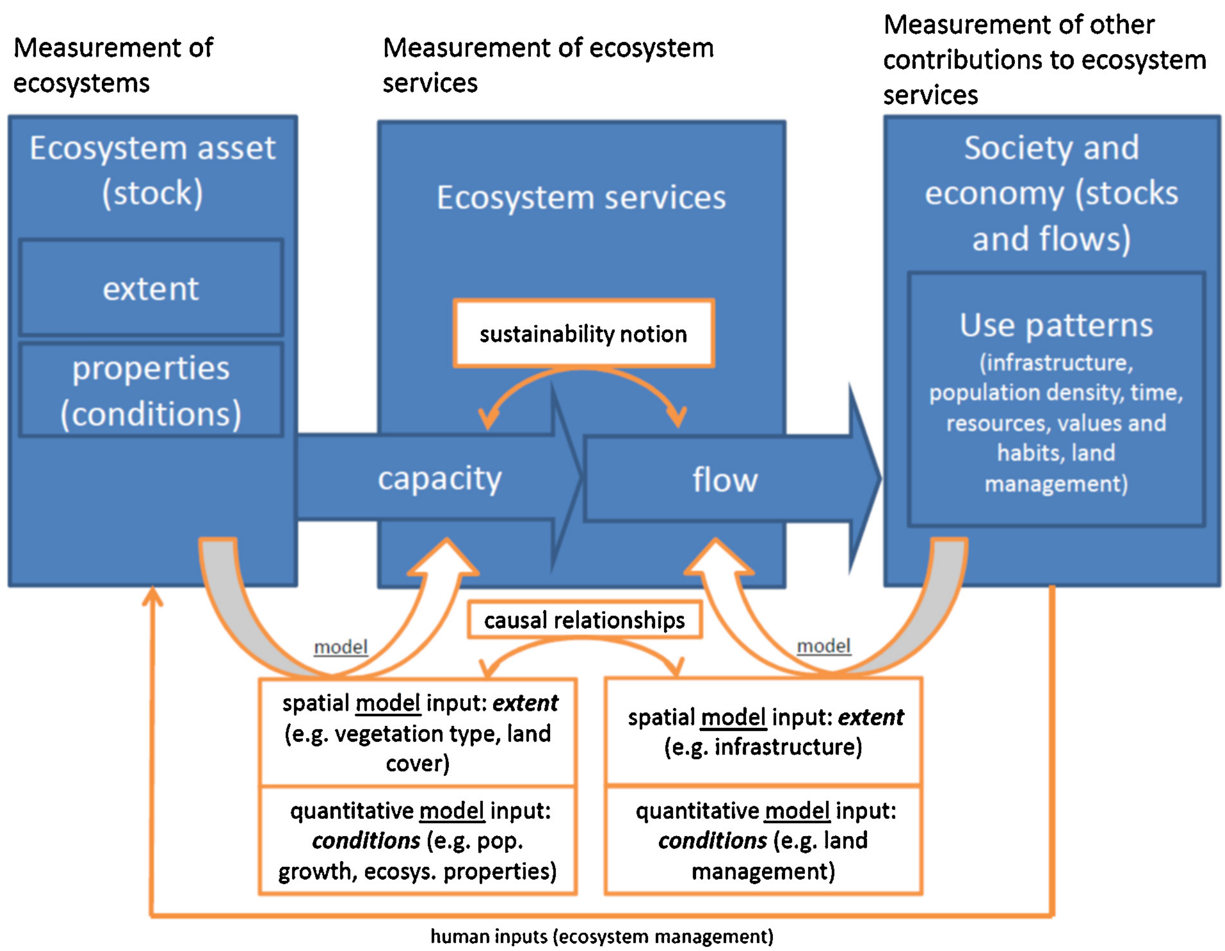

Fig. 1. Integration of ES capacity and flow models in ecosystem accounting.

ES flow is the actual use of ES and occurs at the location where an ES enters either a utility function (of a private household) or a production function (of, e.g. an ES agent) (Schröter et al., 2012). For provisioning services this flow often materialises through some form of extraction (e.g. timber harvest). For regulating services, the capacity is the ability of an ecosystem to modify environmental conditions in a way that is favourable to people (e.g. reduction of flood risks). The flow materialises if people are actually affected by this modification. Cultural services, while being more heterogeneous, often turn into a flow when some form of information is transferred from ecosystems to people (e.g. aesthetic information about the surroundings while hiking).

ES flow differs from ES demand. ES flow is a conceptual idea that focusses on a point in time and space of the last contribution of the ecosystem to human well-being. It is a concept, which contains little or no information about individual agents' preferences for the service, also considering the attributes of potential substitute locations. ES demand is the expression of the individual agents' preferences for specific attributes of the service, such as biophysical characteristics, location and timing of availability, and associated opportunity costs of use. This demand may well be larger than the actual ES flow. For instance, the demand for recreational hiking is covered by substitute locations outside the study region, more carbon could be emitted than can be sequestered within an area, or the risk aversion to snow slides might be higher than the risk reduction that different ecosystems uphill can provide.

Note that capacity and flow as we define it have, in slightly different meanings, been referred to as either supply and demand (Burkhard et al., 2012; Schröter et al., 2012; Tallis et al., 2012) or ecosystem function and service (De Groot et al., 2002; Petz and van Oudenhoven, 2012). However, we think it is worthwhile to distinguish ES specific terms that do not have a different meaning and/or are variously used in economics (Fisher et al., 2008) or ecology (Bastian et al., 2012; Jax, 2005; Wallace, 2007). The distinction between ES capacity and flow has three crucial advantages. First, we gain empirical clarity on the existence of actually used ES versus the potential of ecosystems to provide ES. Second, the distinction between capacity and flow can provide a parsimonious, but policy-relevant and operational indicator of sustainability of human use of ecosystems (cf. Daly, 1977). Third, this distinction is in line with the recently published guidelines for ecosystem accounting (European Commission, 2013).

2.3. Choice of ES

The choice of ES was made to cover a broad range of final, terrestrial ES including provisioning, regulation and cultural services in a Norwegian context (NOU, 2013). We followed the CICES (Common International Classification of ES) scheme version 4.3 (Haines-Young and Potschin, 2013) to categorise the services. It was not possible to cover the whole diversity of ES within one study, therefore nine key ES were chosen. Socio-economic importance of these was indicated through a review of national statistics (SSB, 2012b) and a literature review on land-use in near-natural and cultural landscapes and ES (e.g. Barton et al., 2011; Hytönen, 1995; Kettunen et al., 2013; Moen, 1999). We excluded watershed regulating services. This was partly because of the large expected role of dam regulation of the hydrological cycle compared to the role of abiotic and biotic interactions in providing these services (Barton et al., 2012). Hunting of moose (*Alces alces*) was chosen as moose is a frequently hunted game species in the study area (Helle, 1995). Free ranging flocks of sheep (*Ovis aries*) are an important ES of both forest areas and highland plateaus (Rekdal, 2008). Around 5300 km^2 (about 35% of the case study area) is covered by productive forest. Additionally, around 1600 km^2 (11%) are covered by unproductive forest (with an increment of less than 1 $m^3\,ha^{-1}\,yr^{-1}$) (Eriksen et al., 2006). From these forest areas multiple ES are derived, with both timber harvest and carbon sequestration and storage being two significant ones (De Wit et al., 2006; Hytönen, 1995).

We selected three cultural services that are representative within a Scandinavian context (NOU, 2013). First, the second home (cabin) culture in Norway is a social construct expressing emotional attachment to environmental surroundings (Kaltenborn et al., 2005). Second, we consider recreational hiking, which is the most common outdoor activity in Norway (Jensen, 1995; Vaage, 2009). Third, we include ecosystems without or with low human interference, expressing naturalness of the environment. These areas have been identified to be of high cultural importance in a Norwegian context (Nyvoll, 2012).

The selected ES and their respective indicators are shown in Table 1. ES show different levels of rivalry, i.e. the degree to which their use prevents other beneficiaries from using it (see Table 1). Rivalry is a precondition for creating balances between capacity and flow of ES (Schröter et al., 2012), which is discussed in Section 4.3. All data were, if not indicated otherwise, collected for 2010. All spatial analyses were done with help of ArcMap 10 (ESRI).

2.4. Case study area

Telemark is a county in Southern Norway with an area of 15,300 km^2 and a population of about 170,000 people living in 18 municipalities (SSB, 2012b). Population density varies from about 1 person per km^2 in the west (Fyresdal) and north-west (Vinje) of the county to 65 (Skien) and 176 (Porsgrunn) in the south-east. The altitude ranges from sea level at the coast of the Skagerrak to 1883 m a.s.l. on the Gaustatoppen. The climate varies across the region with temperate conditions in the south-east (Skien, average temperature January −4.0 °C, July 16.0 °C, 855 mm annual precipitation) and alpine conditions in the north-west (Vinje, January −9.0 °C, July 11.0 °C, 1035 mm) (Meteorological Institute, 2012a). With its varied landscape types from fjords to the highland plateau, being representative for the country as a whole, Telemark has been termed “Norway in a miniature”. The landscape is mainly characterised by coniferous and boreal deciduous forest as well as large inland lakes in the southern part, whereas the northern part is characterised by treeless alpine highland plateaus with sparse vegetation (Moen, 1999).

2.5. Description of methods for spatial ES models

2.5.1. Moose hunting

Moose (*Alces alces*) prefers forests and occasionally bogs as habitat, and is to lesser extent present in open and cultural landscapes (Bjørneraas et al., 2011, 2012). To spatially determine the habitat we thus selected the land cover types forest and wooded mires from the national AR 50 land use data set. Moose populations for each municipality were derived from a basic population model based on Austrheim et al. (2011):

$$N_t = Q_t\left\{\left(\frac{C_t - M}{1 - C_t}\right) - (\lambda - 1)\right\}^{-1} \quad (1)$$

where N_t is the post-harvest population, Q_t is the annual harvest (SSB, 2012a), C_t is the pre-harvest proportion of calves in the population (Ungulate register, 2012), M is the natural mortality rate set to 0.05 (Solberg et al., 2012) and λ is the population growth rate calculated as $\lambda = e^r$, where r is the regression coefficient (ANOVA) of the number of seen moose per hunter working day regressed over the years 2001–2010 (Ungulate register, 2012). This coefficient ranged from −0.038 (Kviteseid municipality) to 0.022 (Notodden municipality) (data not shown). The capacity was measured as the recruitment rate of the pre-harvest population $((C_t - M)(N_t + Q_t))$ per km^2 of the selected habitat types) and flow was measured as number of hunted moose (Q_t) per km^2 for the same area.

2.5.2. Sheep grazing

Capacity for sheep (*Ovis aries*) grazing on open alpine and forested summer ranges was modelled with the help of a vegetation map based on satellite imagery (Johansen, 2009) and corresponding assessments of grazing values for specific vegetation types (Rekdal, 2012; Rekdal et al., 2009). These ranged from 0 to 3, with 0 equalling no grazing value, for instance in block fields, 1 equalling moderate grazing value, for instance in heather-rich birch forest, 2 equalling good grazing value, for example in blueberry pine forest, and 3 corresponding to very good grazing value, for instance in grass-rich birch forest. The capacity for the number of sheep grazing per unit of one specific vegetation type was calculated by assigning a conservative estimate of sheep that can be sustained per square kilometre (Rekdal et al., 2009) to each pixel with an assessed grazing value. The capacity model was tested by correlation analysis (Pearson's r) of the log of total capacity (number of sheep km^{-2}) and the log of the sum of satellite-derived net primary production (NPP, in kg C, NASA LP DAAC, 2012) values per grazing area. The flow was measured as the total number of lamb and sheep released minus the number of lost animals per square kilometre for each spatially delineated grazing area (NFLI, 2012).

2.5.3. Timber harvest

Capacity was spatially modelled by using the national land resources dataset (AR5, NFLI, 2010) covering the whole of Telemark under the treeline. Site quality classes, which are classifications to express an area's capacity to produce timber, ranged from 11 (unsuitable), i.e. <1 $m^3\,ha^{-1}\,yr^{-1}$ to 15 (very high), i.e. >10 $m^3\,ha^{-1}\,yr^{-1}$. This spatial information was combined with statistics on annual biomass regrowth ($m^3\,ha^{-1}\,yr^{-1}$) for the region (Telemark, West and East Agder) taken from the most recent national forest inventory (2005–2009) (Granhus et al., 2012).

The flow (harvested timber in $m^3\,ha^{-1}\,yr^{-1}$) was taken from national harvest statistics, where the lowest available resolution was the municipality level (SSB, 2012c) with the assumption that extraction for firewood was at 2005 level, the last year of collection of this data. The flow was delineated with the help of a harvest cost model with harvest costs as a function of accessibility-related terrain-specific costs. This determined areas likely not to be harvested with a positive net yield and thus reduced the area that was determined in the capacity model. The model was developed in a spatially explicit way according to the methods described in Granhus et al. (2011) and consisted of an income layer (timber value) and three cost layers (carriage costs for transportation to the nearest road, cutting costs, and extra costs for steep terrain). Additional costs for a ropeway harvest technique were excluded, as spatial data on where to apply this was missing. The income layer was calculated by multiplying average sale prices for Telemark (SSB, 2013b) with the current harvest mixture of pulp and saw wood (SSB, 2013a). The resulting values were spatially allocated based on the different AR5 site quality classes (NFLI, 2010). Carriage costs were calculated based on path distance to all roads in the county according to a formula given in Dale and Stamm (1994), for roads included in the National road data set (Norwegian Mapping Authority, 2010). Cutting costs were based on standing volume per

Table 1
Overview of selected ES, ES indicators and characteristics (section, division, class after CICES 4.3). For indicator choice see Section 2.5.

Section	Division	Class	ES specification	Capacity indicator	Flow indicator	Rivalry
Provisioning	Nutrition	Wild animals and their outputs	Moose hunting	# recruitment km^{-2} yr^{-1}	# hunted km^{-2} yr^{-1}	Yes
		Reared animals and their outputs	Sheep grazing	# released km^{-2} yr^{-1}	# recaptured km^{-2} yr^{-1}	Yes
	Materials	Fibres and other materials from plants, algae and animals for direct use or processing	Timber harvest	Regrowth m^3 ha^{-1} yr^{-1}	Harvest m^3 ha^{-1} yr^{-1}	Yes
Regulation and maintenance	Maintenance of physical, chemical, biological conditions	Global climate regulation by reduction of greenhouse gas concentrations	Forest carbon sequestration and storage	Sequ. Mg C ha^{-1} yr^{-1} Stored Mg C ha^{-1}	Equals capacity (see Section 2.5/4.2)	Yes
	Mediation of flows	Mass stabilisation and control of erosion rates	Snow slide prevention	Presence of forest land cover on release areas	Presence of forest land cover on release areas if infrastructure in propagation areas present	No
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes	Experiential use of plants, animals and land-/seascapes in different environmental settings	Recreational residential amenity	Capacity (suitability indicator 0–1.0)	Density of cabins km^{-2}	Yes
		Physical use of land-/seascapes in different environmental settings	Recreational hiking	Density of hiking paths km km^{-2}	Density of hiking paths weighted by users	No
	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes	Existence	Existence of areas without technical interference	Areas > 1 km from larger infrastructure as defined by INON	Equals capacity (see Section 2.5/4.2)	No

AR5 site quality class (Dale et al., 1993; Eid, 1998; Granhus et al., 2011). Data on average tree density and standing volume per ha, which was needed for this model, was taken from Eriksen et al. (2006). Extra costs for harvesting in steep terrains were added (Granhus et al., 2011) based on slope data derived from a digital elevation model (DEM).

2.5.4. *Forest carbon sequestration and storage*

Carbon sequestration was modelled as net ecosystem production (NEP), which we calculated as the difference between NPP ($kg\,C\,m^{-2}\,yr^{-1}$) derived from a satellite image (MODIS 17A3, NASA LP DAAC, 2012) and soil respiration (R_s in $g\,C\,m^{-2}\,d^{-1}$) based on an equation from Raich et al. (2002). R was calculated as:

$$R_s = 1.250 \times e^{(0.05452 \times T_a)} \times \left[\frac{P}{4.259 + P}\right] \quad (2)$$

where T_a is the monthly air temperature (1961–1990), and P is the mean monthly precipitation (1961–1990) (Meteorological Institute, 2012b). Soil respiration results were only included when they were not higher than NPP. This means that areas where the difference between NPP and soil respiration was negative were excluded. For instance, areas with little vegetation and low NPP (e.g. bare rocks), but high modelled respiration were excluded because we assumed that not more carbon can be respired than is fixed by plants. We come back to this assumption in the discussion. Carbon removed through harvest was deducted as an average value per municipality ($C\,ha^{-1}$) for the whole forest area. The value was calculated with the help of tree species specific harvest data (SSB, 2012c) and basic wood densities (0.41–0.51) and carbon fractions (0.48–0.51) (IPCC, 2006). The model was tested by calculating Spearman's rho correlation coefficient between the values of the model and a two-layered look-up table (LUT) method based on values for annual carbon sequestration from Framstad et al. (2011). The land cover units in this test model were both tree classes (broadleaf, coniferous and mixed) and site quality classes (as used in the timber model). 100,000 points were set randomly across the study area of which 73,785 could be used for the test.

Carbon storage was mapped with the help of a two-layered LUT based on values for carbon stored ($t\,ha^{-1}$) from Framstad et al. (2011). These were spatially delineated with information on tree classes (broadleaf, coniferous and mixed) and site quality classes (AR5, NFLI, 2010).

As carbon emissions at a global level are by far larger than what ecosystems can sequester all carbon sequestration capacity will constitute a flow. Sequestration and storage capacities by ecosystems will benefit people either in the study region or on a wider (global) scale.

2.5.5. *Snow slide prevention*

We defined the ES snow slide prevention as the contribution of forest vegetation in preventing these slides from taking place. This service was spatially delineated with the help of a snow slide susceptibility model, which was developed to cover the whole of Norway (Derron, 2008). Forest is known to contribute to a reduction of snow slides (Bebi et al., 2001; Brang et al., 2006). Capacity was thus delineated as forest (defined by the AR5 land cover data set, NFLI, 2010), which overlapped with release areas (slope angle between 30° and 55°) of the susceptibility model. Flow only takes place in those release areas that run out into propagation areas of the susceptibility model (Derron, 2008), which contain at least one building from the cadastral dataset (Norwegian cadastral register, 2011) or road infrastructure from the national road dataset (Norwegian Mapping Authority, 2010). This means that for the flow model we excluded those forested release areas that did not contribute to protection because of the absence of beneficiaries that make use of the service.

2.5.6. Recreational residential amenity

Capacity was delineated as suitability for providing a location for second homes (cabins). We analysed choice of location of cabins similarly to geographic species distribution in ecology, namely as a function of environmental variables, using the maximum entropy modelling software MAXENT 3.3.3 (Phillips et al., 2006). As we expected regional differences in habitat choice (motivation for building a cabin), three models were developed: one for coastal cabins (within 1 km from the coastline), one for non-coastal cabins in the proximity of alpine resorts (2 km radius), and one for non-coastal cabins that were not in the proximity of alpine resorts. The first model was run for 4362 presence records of coastal cabins (within 1 km from coastline) from the Norwegian cadastral register. Environmental variables were a DEM, a slope model, Euclidean distance to roads, settlement areas and water bodies, a vegetation type map (Johansen, 2009), and a vegetation type variety map derived from the former. This variety map determined the number of different land cover types for each pixel within a distance of 500 m. The second model was run for 12,254 presence records of non-coastal, non-alpine cabins. Environmental variables were the same as above, with treeline (1000 m a.s.l.) as an additional explanatory variable. The third model was run for 2721 presence records of alpine cabins. Environmental variables were the same as above, with the Euclidean distance to alpine resorts as additional explanatory variable. All three models were combined spatially. The capacity model was tested with the help of area under curve measure of MAXENT (AUC), taking 25% of the input data per submodel as test data. For ES flow we took the presence point density of cabins per km^2 from the cadastral register (27,337 cabins) as an indicator.

2.5.7. Recreational hiking

For modelling capacity we calculated the density of hiking trails ($km\,km^{-2}$) within a search radius of 1 km for the whole county. Hiking trails are registered in the recent national road dataset (Norwegian Mapping Authority, 2010). Taking density as a measure for capacity accounted for the importance of the surrounding of a hiking path. A high density indicated a more developed hiking infrastructure and thus capacity to provide the service. This approach also accounted for accessibility of ecosystems through paths. For the flow we weighted the density of hiking tracks with a combined potential user indicator, which consists of three user groups (local population, tourists, cabin users). In order to combine these three user groups we had to make several assumptions. All hiking tracks within each municipality were given the same weight, assuming that the potential user groups stay within their municipality and use paths equally. There is reason to assume that the likelihood of performing hiking activities is at a comparable level among the different users (Kaltenborn, 1998; Kavli et al., 2009; Vaage, 2009). However, there is little knowledge on when (and where) exactly hiking takes place. We used the following formula to combine the three groups to an potential user indicator x:

$$x = P + \frac{1}{65} \times T + 0.4 \times 3 \times C \quad (3)$$

where P is the number of inhabitants per municipality on 1 January 2010 which was taken from national statistics (SSB, 2012d). T is the number of tourist overnight stays at camp sites, and in cabins, guesthouses and hotels (recreational stays only) in months May to October, which we assumed to be the hiking season. Data was taken from a national tourism database (Statistikknett, 2012) and from the Norwegian Trekking Association (DNT, 2012) for cabins with more than 2000 overnight stays in 2010. Where data was not available for single municipalities but existed only at a higher aggregated level, we took the number of entries in a tourism sector catalogue (Reiselivsbasen, 2012) to proportionally distribute the number of overnight stays to single municipalities. Tourist walking days were calculated as a fraction of inhabitant walking days. One tourist walking day accounts for 1/65 of a local's day. The factor 1/65 results from the assumption that the local population uses 2.5 days per week in the summer half year (26 weeks, i.e. 65 hiking days). C is the number of cabins per municipality as taken from the Norwegian cadastral register (2011). The factor 0.4 results from an average number of days spent in a cabin from May to October, which is about 26 (Kaltenborn et al., 2005) divided by the 65 potential hiking days of the local population. 3 is a conservative estimate of the number of persons per cabin visit (Grefsrud, 2003). The flow model was validated with visitor count data from guest book entries (May–October 2010) of 19 mountain tops spread over six municipalities in south-east and central Telemark (Gundersen, 2013; Hjeltnes, 2012). Pearson's correlation coefficient was calculated to analyse the relation between interpolated values of the flow map and absolute visitor counts at the point of the mountain top.

2.5.8. Existence of areas without technical interference

Capacity and flow of this service are conceptually the same as it is a non-use service as we assume here that existence of wilderness (capacity) implies awareness of and preference for these areas (flow). We used a model of the Norwegian Directorate for Nature Management (1995), which has been adopted for 2008 (Directorate for Nature Management, 2009). The spatial model defines natural areas without technical interference as all areas with a linear distance of more than 1 km distance from existing heavy technical infrastructure. Heavy technical infrastructure includes roads and fortified routes with a length of at least 50 m, railways and power lines as well as regulated water bodies. For a further description of the model see Directorate for Nature Management (2009).

2.6. Spatial analyses

In order to explore the variance of ES capacity and flow values that are found on different land cover units, ES capacity and flow maps where overlaid with the vegetation type map, which determined the spatial ecosystem accounting units (Johansen, 2009). For each vegetation type (land cover/ecosystem functional unit) we calculated the area containing the service and the total quantity as the sum over the range of all 100 m grains (basic spatial units). To test spatial balances of capacity and flow we spatially subtracted flow from capacity layers for two exemplary ES (moose hunting, sheep grazing), while we discuss feasibility of such analyses for the rest of the ES. Balances of absolute quantities of ES were created for timber harvest, moose hunting, sheep grazing and snow slide prevention.

3. Results

3.1. Spatial models

The spatial models of ES capacity are shown for all nine ES in Fig. 2. The northern part of two municipalities in the South-east of Telemark (Skien, Siljan) could partly not be included for three ES (timber harvest, carbon storage, snow slide prevention) as one major spatial input (AR5 land cover data set) did not cover this region.

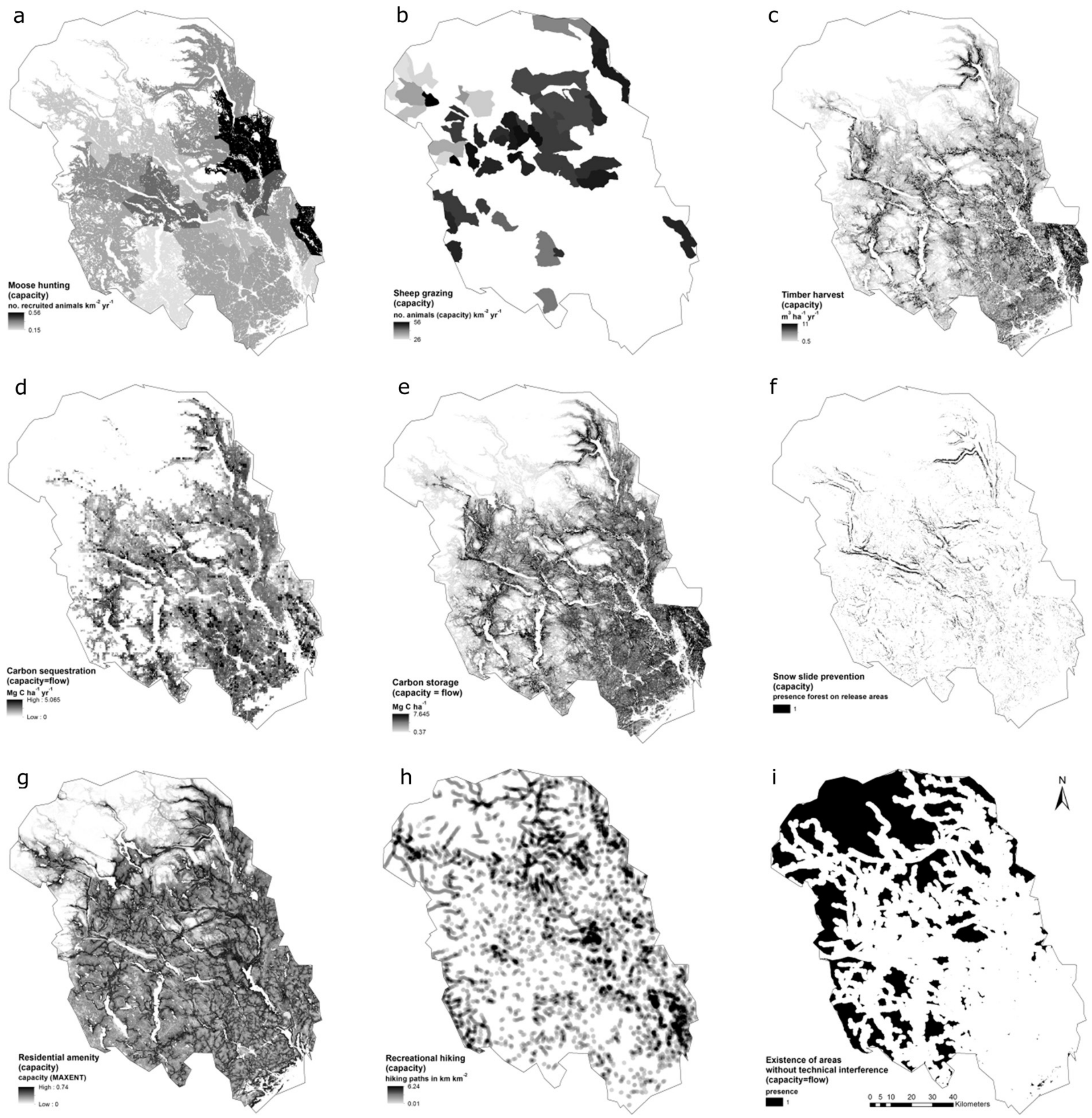

Fig. 2. Spatial models of ES capacity for nine ES in Telemark. White areas indicate that the ES is (per definition) absent. (a–i) Multiple data sources (see Section 2), data access as a member of Norge Digitalt (NINA); (f) Skreddatabase (Norges geologiske undersøkelse); (i) Directorate for nature management.

The resolution of the different services differed depending on methods and spatial data sets used. Three groups of ES models could be distinguished. First, models primarily based on LC and satellite-derived spatial information (timber harvest capacity, carbon sequestration and storage, snow slide prevention, recreational residential amenity capacity) allow for relatively high spatial variability. Second, where such high resolution data is missing, administrative boundaries determine the variation in ES values (LUT approach) (moose hunting, sheep grazing, timber harvest flow). Third, a group of models is primarily spatially determined by human infrastructure (existence, recreational hiking, recreational residential amenity flow).

The spatial models of ES flow are shown in Fig. 3. The services carbon sequestration, carbon storage and existence of areas without technical interference are per definition equal to the capacity models and are thus not shown. Fig. 3 illustrates that ES flow can principally differ from capacity in spatial extent and/or quantities. The services moose hunting, sheep grazing and recreational hiking have the same spatial extent for capacity and flow and differ in quantities. In the case of other services, like timber harvest and snow slide prevention, flow models are a spatial subset of the capacity models because of restricted accessibility.

Spatial ES capacity–flow balances are presented for two example ES in Fig. 4. This spatially delineated quantitative approach gives an indication of the relation between capacity and flow when measured in compatible indicators. It provides information on over- and underuse of the respective service. Estimated moose harvesting rates are slightly above recruitment rates throughout the county except for one municipality (Notodden), which means that flow is higher than capacity and the balances are negative. Except for one small area, capacity for sheep grazing is higher than the flow, which means that vegetation would in principle be able to provide fodder for more sheep (up to 51 animals per km^2 more).

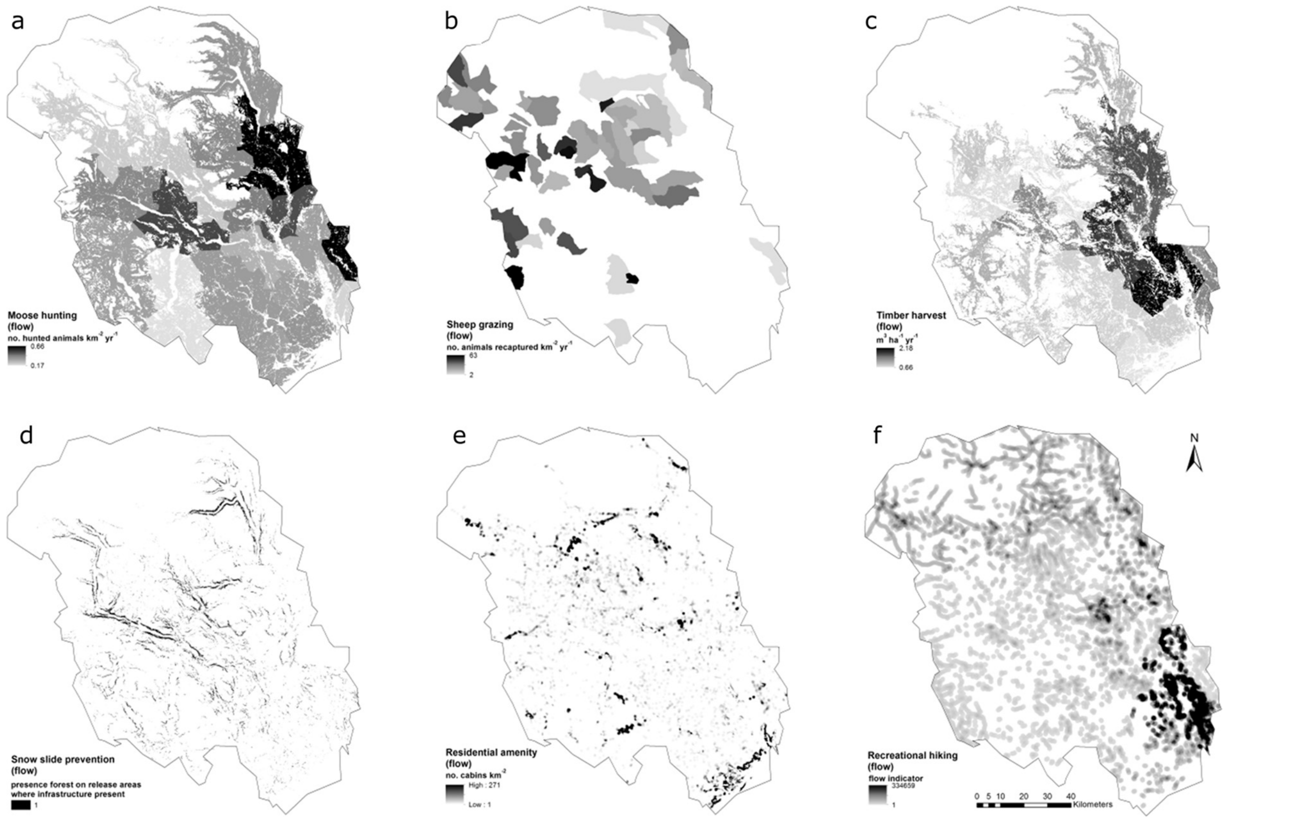

Fig. 3. Spatial models of ES flow for six ES in Telemark. White areas indicate that the ES is (per definition) absent. (a–f) Multiple data sources (see Section 2), data access as a member of Norge Digitalt (NINA); (d) Skreddatabase (Norges geologiske undersøkelse).

3.2. Ecosystem accounting tables

The ecosystem accounting tables in Tables A.1 (capacity) and A.2 (flow) (see annex) show the distribution of ES across 25 vegetation types. The used vegetation map (Johansen, 2009) is the only finer scale land cover map covering the whole county. Certain errors become apparent, for instance that services like timber harvest are allocated to water or agricultural land in this dataset. This is partly due to errors in classification of satellite image or temporal land cover dynamics.

Table 2 illustrates the differences between capacity and flow of ES in absolute figures for the whole county. There is a considerable amount of timber that is not harvested, moose is hunted at a slightly higher rate than the annual recruitment rates, the capacity of sheep grazing is much larger than the actually used flow of the service and, finally, snow slide prevention is in principle provided but not used in the sense of protected infrastructure on more than 8000 hectares.

3.3. Validation results

Four models to map ES were validated. Others could either not be tested as all available data was used to build the spatial model (timber harvest, moose hunting) or as they were defined

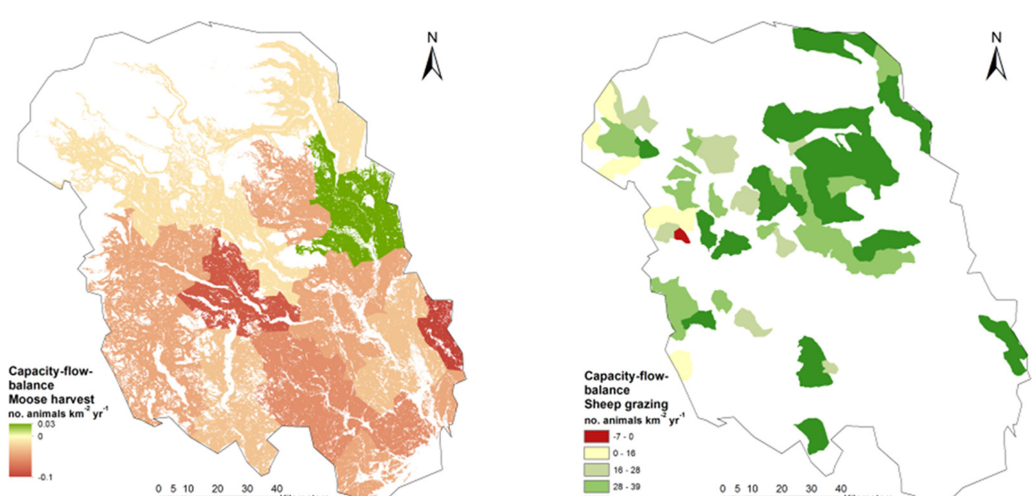

Fig. 4. Capacity–flow-balance for two example ES (Moose hunting and sheep grazing).

Table 2
Absolute capacity–flow-balance per vegetation type for four selected ES.

Vegetation type (ecosystem functional unit)	Timber harvest		Moose hunting		Sheep grazing		Snow slide prevention
	Area (ha)	SUM (m^3 yr^{-1})	Area (ha)	SUM (# animals yr^{-1})	Area (ha)	SUM (# animals yr^{-1})	Area (ha)
Coniferous forest (dense)	55,698	541,671	0	−70	0	22,373	3101
Coniferous and mixed forest (open)	40,427	277,715	0	−41	0	13,961	1403
Lichen rich pine forest	16,732	59,428	0	−11	0	1893	590
Low herb broadleaved forest	17,399	121,435	0	−17	0	10,116	491
Tall-fern and tall-herb broadleaved forest	6287	74,220	0	−9	0	3784	255
Bilberry birch forest	55,483	281,674	0	−42	0	23,211	1261
Cowberry birch forest	10,808	34,537	0	−6	0	3094	308
Lichen rich birch forest	7888	26,708	0	−5	0	1342	267
Ombrotrophic hummock and lawn bog	7300	11,695	0	−2	0	3283	25
Rich lawn fen	5617	7939	0	−2	0	2577	15
Rich mud-bottom fen	1866	7613	0	−1	0	694	22
Alpine ridge vegetation and barren land	791	3255	0	0	0	1345	17
Graminoid and wood-rush ridge	369	1083	0	0	0	1220	14
Heather rich alpine ridge vegetation	2586	3843	0	−1	0	7906	40
Lichen rich alpine ridge vegetation	10	6	0	0	0	900	0
Early snow patch vegetation	2153	9388	0	−1	0	6294	34
Alpine heather and dwarf birch heath	10,947	22,465	0	−4	0	22,720	174
Alpine fern meadow	2261	9350	0	−1	0	8718	20
Grass and dwarf willow snow patch	1042	1822	0	0	0	1445	36
Poor bryophyte snow patch	2589	4685	0	−1	0	3453	62
Glacier and snow	1	5	0	0	0	50	0
Water	9879	37,198	0	−2	0	6493	143
Agricultural land	4115	40,854	0	−1	0	963	11
City, densely populated areas	600	3766	0	0	0	12	1
Unclassified	193	519	0	0	0	118	4
SUM	263,041	1582,873	0	−221	0	147,963	8294

by empirically measured spatial input data (existence, recreational residential amenity flow, recreational hiking capacity). The sheep grazing model showed a strong correlation ($r = 0.885$) with satellite derived NPP data. The forest carbon sequestration model showed rather weak relation ($r = 0.339$) with the chosen validation model (LUT). The accuracy of the recreational residential amenity model showed a good ability to predict suitability for the sub-models that are close to the coast (AUC = 0.844) or close to alpine resorts (AUC = 0.892). The predominant part of the county (non-coast, non-alpine) was characterised by a lower, but acceptable model quality (AUC = 0.682), which was distinct from random distribution (AUC = 0.5). The recreational hiking model showed a strong correlation ($r = 0.786$) with visitor data.

4. Discussion

In this section we highlight some of the challenges of modelling ES capacity (Section 4.1) and flow (Section 4.2). Based on that we discuss conditions that necessarily need to be fulfilled in order to create meaningful spatially explicit balances between ES capacity and flow (Section 4.3). Furthermore, we examine the contribution of spatial ES mapping to creating ecosystem accounting schemes (Section 4.4).

4.1. Modelling capacity

Several spatial and non-spatial data-sets were used to generate the different ES capacity models which we discuss in detail below. For moose hunting, our approach does not consider habitat connectivity, local hot spots or avoided habitats as has been done in other studies on a smaller scale with access to radiometric data (Bjørneraas et al., 2012; Dettki et al., 2003). Given richer data access, however, capacity could also be understood as the capacity of vegetation cover to provide forage for moose (e.g. young stage of broadleaf trees, blueberry cover, herbs (cf. Solberg et al., 2012)). The indicator would then move down one trophic level in the food chain, quantifying primary production instead of primary consumption.

Both the timber capacity model and the carbon storage model combine spatially explicit estimations of the site quality class with recent measurements. This so-called LUT approach has frequently been applied in ES mapping studies (Martínez-Harms and Balvanera, 2012). While such an approach allows for coverage of large areas, quantitative differences within the single classes are not considered, so that this method is necessarily a simplification. The satellite-derived method for modelling carbon sequestration is able to cover the whole region in the absence of field data. The method is comparable to other large-scale ES studies (Raudsepp-Hearne et al., 2010) but further elaborates these as it includes respiration next to NPP. We had to restrict the model to forested areas as modelled respiration was much higher than actual NPP in the northern regions of the county. Here, absence of soil and harsh climatic conditions limit NPP. There is reason to assume that non-forested areas in Norway have a neutral carbon balance (Grønlund et al., 2010), which is why we neglected these areas for this assessment. The model showed relatively low correlation with the LUT validation model. This might partly be due to the fact that MODIS NPP data are aggregated over large areas (1 km by 1 km grain size) whereas the validation model consists of higher resolution land cover maps.

The snow slide prevention model is a spatially explicit binary LUT, which assumes that if forested vegetation is present on slopes susceptible to snow slides, the capacity is present. If this coincides with infrastructure and buildings in the slide area the flow is delivered. Forested areas have been accounted for in large-scale mapping of avalanche susceptibility before (Barbolini et al., 2011). Such an approach does, however, not account for different qualities of forests, e.g. tree densities, age, that might influence the actual ability to prevent snow slides. However, such data collection would require extensive field work which was not within the scope of our study.

The recreational residential amenity model assumes that suitable locations of cabins can be derived by the presence of existing cabins. In reality, the location of new cabins might primarily be determined by the land owner's and municipality's decision to allow for development of an area into a cabin site. The results of the three spatial sub-models, however, showed a fair to strong ability to predict the presence of a cabin with the available data.

The recreational hiking capacity model is based on the assumption that hiking takes place on hiking trails and their surroundings. This restricts capacity to ecosystems that have been changed, i.e. made accessible through trails. In principle non-accessible areas also provide capacity for this ES. Other studies have included such areas irrespective of whether they are accessible or not. Raudsepp-Hearne et al. (2010) have used forest cover as a whole as an indicator for recreation, while Haines-Young et al. (2012) and Burkhard et al. (2012) give weights to different land cover types. In contrast, our approach considers actual accessibility and thereby allows for more spatial variability. Our model also assumes that all areas with a hiking path are equally aesthetically attractive for hikers. This is of course not the case in reality. Many of the hiking paths in Norway are based on old transport routes, which were not constructed based on aesthetic or recreational preferences. Data on landscape preferences, however, was incomplete or ambiguous (Gundersen and Frivold, 2008), and spatially explicit data to build a more informed model unavailable.

4.2. Modelling flows

One type of flow models that we used delineates statistical harvest data with the help of spatial information derived from the capacity models (moose hunting, sheep grazing). For the service timber harvest the potential flow area was constrained by taking costs of access into account. In principle, even single trees on unproductive sites far from forest roads can be harvested to realise a flow. This, however, is unrealistic as access costs are too high. Our model, which is a spatially explicit version of a tested forestry approach (Granhus et al., 2011), accounts for terrain in which forest grows and is harvested. The flow model thus forms a spatial subset of the capacity model, excluding areas that are accessible only at high economic costs and where beneficiaries are likely to be absent. The latter condition is based on the requirement that for a flow the presence of a beneficiary is needed (Schröter et al., 2012). Two other flow models also constitute a spatial subset of the respective capacity models, namely snow slide prevention and recreational residential amenity. In the snow slide prevention model we included only those forest areas, which protect areas where beneficiaries are actually present. It is important to note that the assumption in the flow indicator is that 100% of avalanche risk is removed with forest vegetation. Avalanche risk avoidance perceived by the population could potentially be formulated as an ES demand and may exceed the flow (risk avoidance actually provided by vegetation). We have also assumed that all release areas are evenly prone to snow slides irrespective of actual snow precipitation in the respective year. The recreational residential amenity model shows areas that are not only suitable but in fact used as a location for cabins. It assumes, however, that cabins are evenly in use, which in reality is not the case as some are empty and others are more frequently used. The recreational hiking flow model follows a slightly different approach. Here, actual presence of beneficiaries determines the quantity of the flow. This model inherently assumes that people hike in the wider surroundings of a cabin (as defined by municipal borders), a tourist accommodation or their homes. This is a simplifying assumption that costs of access (i.e. travel costs) increase beyond the municipality's border. The validation result of this model, however, exhibits a strong correlation with visitor data. The assumption that for carbon sequestration and storage flow equals capacity is derived from the observation that certain ES have beneficiaries across different spatial scales (Hein et al., 2006). Under current greenhouse gas emission status, there would be beneficiaries outside Telemark even if the county's forests would be able to sequester more than the local emissions. The latter is not the case, as greenhouse gas emissions of Telemark are at about 4.3 million tonnes CO_2 equivalent (Fylkesmannen i Telemark, 2008) which means that the total estimated sequestration (1.05 million t C, equalling 3.85 million t CO_2) accounts for 89.6% of what is emitted. Existence of areas without technical interference was taken as an indicator for wilderness-like areas that people might attach existence values to. In our conceptual model, we consider that flow is effective information about the capacity areas. With this flow indicator we assumed that all capacity areas are known to the public.

In order to empirically reflect long-term sustainability of ES flows further aspects would need to be considered. This would include going beyond, for instance yearly extraction and comprise aspects of maintenance of biodiversity and resilience of ecosystems. In the light of high data needs, however, this seems ambitious to express and analyse with the help of suitable spatial indicators. Furthermore, a conceptualisation that builds on sustainable yield of ecosystems, might neglect the crucial environmental-ethical question about how much of an ecosystem's capacity should be available for direct human use and how much for non-human purposes.

4.3. What is needed to analyse a spatial capacity–flow-balance?

Creating spatial balances between ES capacity and flow has recently drawn increased research interest (Burkhard et al., 2012; Nedkov and Burkhard, 2012). Such an approach basically subtracts flow from capacity per spatial unit and can be used to analyse the sustainability of ecosystem use. Several questions arose on what is required in order to create meaningful capacity–flow-balances (Schröter et al., 2012). We have identified five conditions for creating such a balance, which we discuss below and which are all met by the two examples shown in Fig. 4. All other ES in our case do not fulfil at least one of the conditions.

First, a conceptual difference between capacity and flow is needed. For a metaphysical service like existence of areas without disturbance, capacity and flow are in our case per definition equal because the value lies in the capacity being physically unaltered. If people hold an immaterial non-use value for largely undisturbed ecosystems, then the capacity and flow should be equal.

Second, spatial delimitation of both capacity and flow needs to be possible (Schröter et al., 2012). In the case of carbon sequestration and storage we have argued that given current global carbon emission levels, all of the service's capacity is actually used. Given the (theoretical) case that this does not apply, it would be impossible to spatially determine which areas in fact provide the ES flow used by a specific group of beneficiaries and which do not, as carbon is distributed in the atmosphere. It cannot be pinpointed where the carbon emissions of these beneficiaries are fixed.

Third, capacity and flow should have the same spatial extent. We have argued elsewhere that flow should be mapped at the place of the last contribution of the ecosystem (Schröter et al., 2012). However, in the case of the service timber harvest, flow is the use of a long-time aggregate of the capacity (yearly increment). Flow thus takes place locally, and once in 80–100 years, i.e. within a short time frame relative to the ecological processes involved. While comparing these two values aggregated for a whole county gives an informative estimate of how much of the annual capacity is actually used, a spatial balance would require defining either spatial or temporal aggregations. In the first case spatial sub-regions that average the annual harvest and regrowth would need to be delineated. In the second case, a temporal assumption would be required of how capacity of each basic spatial unit adds up over the time period needed to build a harvestable stock.

Fourth, ES need to be rival or congestible (cf. Table 1 and Schröter et al., 2012) as a balance presumes depletion. Both snow slide prevention and recreational hiking are non-rival, i.e. their use does in principle not prevent other beneficiaries from using it. However, such services can be characterised as congestible when they are non-rival up to a certain threshold of use intensity beyond which additional users will subtract from the benefits to existing users (Kemkes et al., 2010). For congestible services a capacity–flow-balance is thus reasonable if the use threshold can be defined. This remains a challenge for further research. For instance, the number of people that could hike at the same time in a given area or the number of houses that can be built in a valley protected from snow slides by a forest would need to be determined either theoretically or empirically by asking current users. Policy choices will have to be made about use levels also for rival services, such as recreational residential amenity. A higher use of possible locations for cabins might lead to environmental problems including a disruption of natural scenery.

Fifth and finally, capacity and flow need to be measured with similar indicators so that units can be subtracted. For the service recreational residential amenity this would require transferring capacity, which is expressed here as suitability into an indicator similar to the flow indicator (cabins per km^{-2}). Information on a maximum socially accepted density of cabins in suitable areas would be needed.

These conditions could be met by most provisioning services. For most regulating services, it seems that providing maps of both capacity and flow is useful, but creating a balance between them is not suitable. The group of cultural services is more heterogeneous, and, as we have discussed, some might meet all criteria.

Further research in the emerging field of mapping ES should focus on the development of suitable indicators for capacity and flow, which are compatible. Furthermore, spatial delineation of services, in particular of cultural services, needs further advancement. An important question that remains to be explored is the question, which effect over- or underuse of one respective ES has on the state of other ES.

4.4. Spatial ecosystem accounting

The accounting tables for ES capacity (Table A.1) and flow (Table A.2) provide a first step towards ecosystem accounting. The SEEA ecosystem accounting guidelines discuss the need for measuring the extent and condition of ecosystems as well as monitoring ES (European Commission, 2013). The work presented here focuses on the latter aspect, but extent and properties of ecosystems form an inherent part of several ES capacity models (Fig. 1). We argue that the two-sidedness of ES (capacity and flow) provides relevant information on sustainable use of ecosystems and should therefore be monitored. Including balances between capacity and flow (Table 2) in an accounting system can show the difference between the full potential of ecosystems to provide final services and the current use of it.

A spatially explicit approach, also recognised by SEEA (European Commission, 2013), enables monitoring and expressing changes in land-use for a basic spatial unit in ecosystem accounting schemes through changes in extent and characteristics of ecosystems which determine ES capacity. Such land-use changes might also change ES flows if the basic spatial unit is the site of an actually used ES, prior to the change. ES flows depend on socio-economic factors, as we showed in our models (e.g. population density, infrastructure). As an example, a change in socio-cultural contributing factors to ES provision, e.g. the increase of tourist overnight stays in a region, could lead to an increase in ES flow, while capacity to provide the ES recreational hiking stays the same. For ecosystem accounting, this would mean not only systematically monitoring ecosystem inputs into models, but also socio-economic data in a spatially explicit way. Relevant socio-economic factors include, but are not limited to population densities or densities of infrastructure per spatial unit.

5. Conclusion

The objective of this study was to test and validate spatial models of ES capacity and flow. We have demonstrated that a careful conceptual definition and choice of suitable indicators is needed for spatial assessments of ES. We have shown that combining a set of spatial modelling methods presents an opportunity to distinguish capacity and flow of ecosystem services at a large scale. Such models can support ecosystem accounting by allocating statistical ES values to spatial accounting units. These values can be derived with the help of a variety of mapping methods which include (multiple layer) look-up tables, causal relations of datasets (e.g. satellite images), environmental regression and indicators derived from direct measurements.

We have empirically shown that ES capacity and flow differ both in spatial extent as well as in absolute quantities. Access to areas that exhibit an ES capacity involves costs (e.g. harvest costs and travel costs to distant ecosystems), which can predict whether a beneficiary (ecosystem manager, private person) is actually present. Consequently, such spatial constraints can create ES flow models that are spatial subsets of the capacity models. Hence, the case of spatial accessibility also challenges the assumption that biophysical mapping without considering economic costs and benefits is possible for all ES.

Furthermore, quantities of ES flow per unit area can be higher or lower than ES capacity. Maps of balances between ES capacity and flow have the potential to inform policymakers about over- or underuse of the respective service in a spatially explicit way. Spatial balances between capacity and flow are mainly applicable for provisioning services which satisfy the condition of rivalry. For other services, such as many cultural services, indicators and use thresholds need to be defined properly before a spatial balance between capacity and flow can be created. Such methodological advancements are a critical element to understanding spatial patterns in the sustainability of ecosystem use, and for developing ecosystem accounts.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolind.2013.09.018. These data include Google maps of Telemark county.

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