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Operationalizing the ecosystems approach: Assessing the environmental impact of major infrastructure development

Joanna E. Zawadzka[a], Ron Corstanje[a,*], Jacqueline Fookes[b], Jonathan Nichols[b], Jim Harris[c]

[a] Cranfield Soil and Agrifood Institute, Cranfield University, Cranfield, Bedfordshire MK43 0AL, United Kingdom
[b] Mott MacDonald,69-75 Thorpe Road, Norwich NR1 1UA, United Kingdom
[c] Cranfield Institute for Resilient Futures, Cranfield University, Cranfield, Bedfordshire MK43 0AL, United Kingdom

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ABSTRACT

The ecosystem services approach is increasingly applied in the context of environmental resources management and impact assessment. Assessments often involve analysis of alternative scenarios for which potential changes in ecosystem services are quantified. For such assessments to be effective there is a requirement to represent changes in ecosystem services supply in a clear and informative manner. Here we compute Ecosystem Services Ratio (ESR), a simple index that quantifies the relative change in ecosystem service provision under altered conditions given the baseline, and the Modified Ecosystem Services State Index, which averages the ESR scores obtained for each ecosystem service assessed, to provide an overall measure of the change. Given that modelling approaches to quantification of ecosystem services often result in production of maps of ecosystem supply, the proposed metrics can be visualized as maps in support to decision making processes. We use these indices to investigate potential changes in the supply of seven modelled ecosystem services resulting from the introduction of a major road development – a highway with associated green infrastructure – into a predominantly agricultural landscape in the UK. We find that the planted woodland, scrubland and grassland can increase the supply of multiple ecosystem services not accounted for in previous green infrastructure studies, although the magnitude of change differs with the type of vegetation, initial conditions and timeframes of the assessment.

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1. Introduction

The concept of ecosystem services, generally defined as the benefits that humans obtain from nature (Millennium Ecosystem Assessment, 2005), has introduced a new dimension to the environmental discourse, that of natural capital. The concept of natural capital has stemmed from the realization that there is a finite capacity of utilization of the natural environment, beyond which environmental degradation results in the loss of ecosystem services essential to human and societal wellbeing. Traditionally, approaches to development planning have focused on the "grey" or hard infrastructure with effects on blue (water) and green infrastructure having been given secondary consideration. The conceptual aspect of natural capital in the ecosystem services approach opens up the opportunity for not only mitigating the potential negative environmental impacts but also optimizing the development to maximize the flow of ecosystem services from the affected area. This offers progress towards integrating "grey-green-blue" infrastructure on an equal footing, recognizing the constraints and opportunities for investment aimed at optimizing the outcome of the development, with net gains in natural capital.

Environmental impact assessments (EIAs), defined as "the evaluation of the effects likely to arise from a major project (or other action) significantly affecting the environment" (Jay et al., 2007) can provide a mechanism for inclusion of the green and blue infrastructure into development design as a means for environmental mitigation. The use of EIAs was advocated by the Agenda 21 (UNCED, 1992) document with the aim of promoting sustainable development in making integrated energy, environment and economic policy decisions. Other potential benefits of EIAs include providing an aid to decision-making and to the formulation of development actions, as well as a vehicle for stakeholder consultation and participation (Glasson et al., 2012). Despite the widespread application of the EIA process multiple issues that prevent the EIA to fully serve its substantive purpose have been identified. For example, (Cashmore et al., 2007) reviewed three case studies where

* Corresponding author.
E-mail address: roncorstanje@cranfield.ac.uk (R. Corstanje).

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an EIA was applied with the aim of empirically assessing their possible short term (less than five years) contributions to sustainable development. They found that the role of the EIA's impact on the design phase was limited, which could mostly be attributed to "fine-tuning, rather than influencing the selection of strategic alternatives" and that the length of documentation provided by the EIA prevented the report from being the primary source of information on which the consent was based. Conversely, Baker et al. (2013) recognized strengths of the ecosystem services approach as applied ex-post to several EIA case studies. These included better integration of particular topics of the assessment via consideration of ecosystem services bundles allowing for identification of synergies and tradeoffs between ecosystem services at various spatial and temporal scales (Raudsepp-Hearne et al., 2010; Rodríguez et al., 2006), and the framing of the entire assessment in a more tangible way for stakeholders and decision makers by treating the environment as a source of benefits towards social and economic goals.

The stocks and flows of ecosystem services can be enhanced by introduction, maintenance or ecological restoration of existing green infrastructure (GI) in any major infrastructure development. GI is often defined as "the network of semi-natural landscape elements" (Opdam et al., 2015), within the area affected by the new development. There are multiple examples in literature where ecosystem services of existing green infrastructure have been analyzed or quantified, particularly in urban environments in the context of abating the air and noise pollution associated with urban development and road transport (Gratani and Varone, 2013; Pugh et al., 2012; Silli et al., 2015). As so far, examples of ecosystem services assessments conducted for planned green infrastructure are few. We are addressing this gap by presenting a case study whereby the ecosystem services approach was applied to assess the short to long term ecosystem value of green infrastructure (planted woodland, scrubland, and grassland) planned in the design of a major road development accepted for construction in the UK. The specific objectives of this paper are therefore to: (1) present a quantitative method for analyzing changes in ecosystem services supply in the context of the environmental impact of a planned development forecast over four milestone years of the development design, (2) illustrate changes in ecosystem services in a clear, informative and spatially unambiguous manner, (3) discuss the impact of green infrastructure introduced with the development on the selected ecosystem services, and (4) demonstrate the potential implications of consideration of land use transitions during the ecosystem services assessment on the development design. We hypothesize that answering these objectives will identify the potential benefits of conducting spatially explicit ecosystem services assessments in selected aspects of development planning and impact assessment, and emphasize the importance of inclusion of green infrastructure for environmental mitigation and enhancement purposes.

2. Materials and methods

2.1. Study area

Our case study is a stretch of approximately 20 km of a planned highway with associated landscaping located north of Norwich in East Anglia, UK (Fig. 1) and covers the area of 371 ha defined by the Development Consent Order boundary. The main purpose of the road is to provide a northern overpass of the city that would reduce existing congestion of traffic on roads that are unsuitable for the current traffic levels (The Norfolk County Council, 2014). The environmental setting of the road includes mostly arable land dominated by wheat and oil-seed rape cultivation, with addition of pastures and patches of coniferous and broadleaved forests. The settlements of Norwich are located to the south of the road at a distance of approximately 100–3000 m and further scattered settlements are located to the north of the proposed road at a distance ranging between approximately 100–400 m. The road cuts through the northern edges of the Norwich Airport located north of the settlements of Norwich. The soils are sandy or loamy and the average annual temperature and precipitation are 10.5 °C and 652 mm respectively.

The GI introduced by the development is planned to be composed of native plant species and will cover approximately 176 ha, constituting 47.5% of the total study area. The most extensive GI class is grassland (114 ha), followed by woodland (57 ha) and scrubland (5 ha), and these will be planted on existing arable land (122.5 ha), pastures (29 ha), forests (8 ha) and various other manmade land use types (16 ha). The planned road, covering approximately 70 ha (19% of the study area), will be introduced on existing arable land (40 ha), other roads (12 ha), pastures (9 ha), airport (4 ha), forests (2 ha), and other land uses (2 ha). Remaining new land uses such as lagoons, ponds, and paths will cover 8.5% of the study area (31 ha), and no directly induced change is expected on 25% (93 ha) of the area.

2.2. Quantification of ecosystem services supply

Selected ecosystem services (Table 1) were assessed within four milestone years which corresponded to the milestone design years used in Environmental Impact Assessment (The Norfolk County Council, 2014) conducted for the scheme. These included the year 2012, referred to in this study as the baseline year, the opening year of the road (2017), the design year – 15 years of opening (2032), and the design life of the scheme – 60 years of opening (2077), referred to as the future conditions or states for which changes in ecosystem services are compared to the baseline. Since our analysis was performed in the context of the EIA, we adhered to the milestone years predefined in the EIA, and assessed ecosystem services that corresponded to the topics of the EIA. Our intention was to perform a spatially-explicit assessment, and therefore our choice of ecosystem services was limited by the availability of modelling tools that worked in spatial domain. Additionally, due to the significance within the study area, we assessed changes in agricultural production based on literature-informed yield projections for major crops cultivated in the area.

The biophysical indicators of selected ecosystem services for each milestone year were quantified using the Integrated Valuation of Environmental Services and Trade-offs (InVEST) (Tallis et al., 2013) modelling suite version 2.6.5 developed by the Natural Capital project. InVEST had been designed to provide tools for quantitative assessments of distribution of ecosystem services across landscapes allowing for spatial recognition of differences in the supply of ecosystems services. This is achieved by using land use/land cover maps as primary modelling inputs, which are assumed to be the main drivers of change in ecosystem services. Estimating the state of ecosystem services in various steps in time required that the key modelling parameters that are expected to change over time, such as projected climatic conditions or increased amounts of biomass due to vegetation growth, were adjusted accordingly for each milestone year. In particular, carbon sequestration modelling was driven by attributing expected carbon stocks in soil, above-, below-ground and dead biomass for each land use class present in the study area, and in our case we made account for the incremental increase in carbon stocks due to sequestration to soil and in growing biomass over the milestone years. The sediment retention model calculates avoided soil loss by land use compared to bare soil and is based on the revised USLE soil erosion model (Wischmeier and Smith, 1978), whose parameters were adjusted accordingly to changing land use, topography

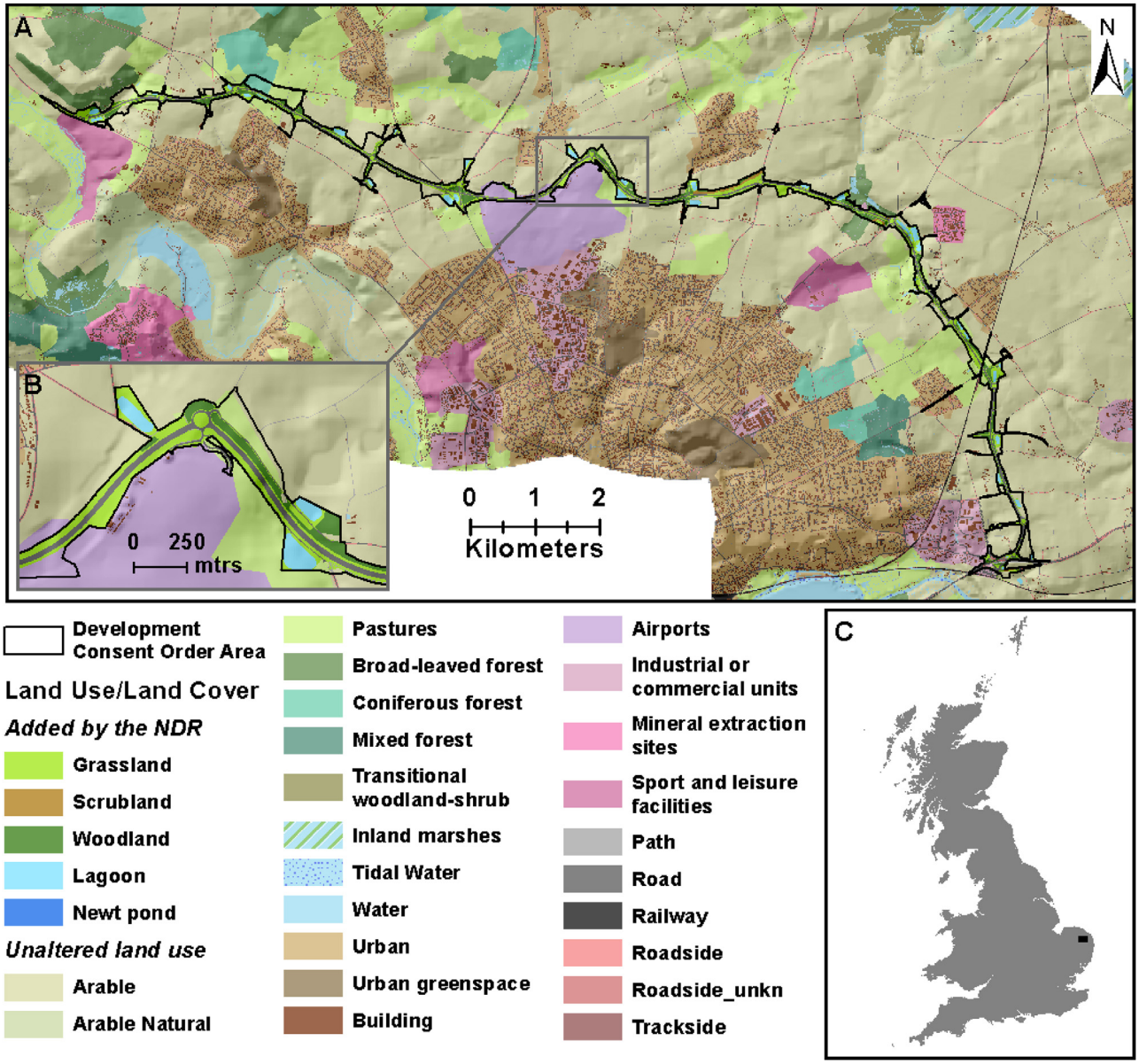

Fig. 1. Map of the study site representing: A) the current land use/land cover and planned development (the NDR – Northern Norwich Distributor Road), B) a close-up to the fragment of the study area depicted in Fig. 2, and C) the location of the study area within Britain.

Table 1
Selected ecosystems services and methods of assessment used in this study.

Ecosystem service type	Ecosystem service	Method of estimation	Units	Corresponding topics of the Environmental Statement
Supporting services	Avoided habitat risk	Based on the InVEST 2.5.6 Habitat Risk Model output	Unitless	Nature conservation
Regulating services	Carbon storage and sequestration	InVEST 2.5.6	Mg ha^{-1}	Carbon
	Sediment retention	InVEST 2.5.6	Mg ha^{-1}	Geology and Soils
	Nitrogen and Phosphorus retention	InVEST 2.5.6	kg ha^{-1}	Road drainage and the water environment
Provisioning services	Arable production	Literature based estimation	Mg ha^{-1}	Community and private assets
	Hydropower production (water yield)	InVEST 2.5.6	mm	Road drainage and the water environment

represented by a 10 m resolution digital elevation model (DEM), and climatic conditions over time. The water yield model determines the amount of water running off each grid cell of a DEM as the precipitation less the fraction of the water that undergoes evapotranspiration, and was parameterized with different climatic values and changes to topography over the milestone years. The nutrient (N and P) retention models estimate the amount of nutrients retained within the landscape based on the modelled amount of water running off each grid cell together with nutrient loadings for each land use type and nutrient retention efficiency of each land

Table 2
Interpretation of the values of the Ecosystem Service Ratio and Modified Ecosystem Services Status Index.

ESR/MESSI value	Legend entry
0	No service
0–0.1	Over 10 times reduction
0.1–0.2	10 to 5 times reduction
0.2–0.5	5 to 2 times reduction
0.5–0.67	2 to 1.5 times reduction
0.67–1.00	Less than 1.5 times reduction
1	No change
1–1.5	Up to 1.5 times increase
1.5–2	1.5 to 2 times increase
2–5	2 to 5 times increase
5–10	5 to 10 times increase
10–100	10 to 100 times increase
>100	Over 100 times increase

use class within the study area, and were adjusted to make account for different climatic conditions expected in each milestone year. Finally, the habitat risk model uses land use maps for indication of locations of landscape features considered as threats for land uses providing habitats for biodiversity. Distance between these features as well as scores for exposure to threats – in our case the new road – and their consequences were used to assess habitat risk, and in this case these parameters were assumed stable across all milestone years.

Each InVEST model deployed in this study generated 10 m resolution gridded raster maps with the magnitude of the biophysical indicator for a given service assigned to each grid cell and these maps were subsequently used to compare the state of ecosystem services between the baseline and future conditions. Additionally, due to its significance within the study area, the provisioning service of arable production was estimated from projected yield changes of wheat, sugar beet and rapeseed oil in the future (Jaggard et al., 2010), taking the average yield of years 2000–2010 as the baseline.

2.3. Quantification of changes in ecosystem services supply

Changes in the supply of selected ecosystem services between the baseline and future conditions were quantified using the Ecosystem Services Ratio (ESR) index (Eq. (1)). The ESR assesses the relative change in the supply of a single ecosystem service between the baseline and future conditions, with 1 indicating no change, values between 0 and 1 – a decrease in the supply, values over 1 with no upper limit – an increase in the supply, and 0 – a complete loss of that service (Table 2).

$$ESR = \frac{ES_{future}}{ES_{baseline}}, \quad (1)$$

where ESR – ecosystem services ratio; ES_{future} – the absolute amount of units of ecosystem service in the future state (i.e. after implementation of the project) and $ES_{baseline}$ – the absolute amount of units of ecosystem service in the baseline state (i.e. before implementation of the project).

The overall change in the supply of multiple ecosystem services was assessed with the Modified Ecosystem Services Status Index (MESSI). MESSI is calculated as a mean of the ESR values for each ecosystem service compared between the given set of baseline and future condtions (Eq. (2)) and therefore assumes the same ranges of possible values as the ESR. MESSI is based on the Ecosystem Services Status Index (ESSI) (Leh et al., 2013) that quantifies only the portion of ecosystem services gained or lost with respect to baseline values.

$$MESSI = \frac{\sum ESR_i}{n}, \quad (2)$$

where MESSI – Modified Ecosystem Services Status Index; ESR_i – ecosystem services ratio for service i in a given time step; n – number of ecosystem services assessed.

It is important to note that the biophysical indicators of ecosystem services used in the calculation of the ESR index should follow the rule that the higher value of the indicator corresponds to the more desirable state of the ecosystem service it represents. The habitat risk model output map did not follow this rule as it quantified a negative state of ecosystem services, and therefore had to be modified prior to calculation of the ESR. Since the potential maximum value of habitat risk was known, "avoided habitat risk" indicator map was generated by subtraction of the modelled habitat risk values from the known potential maximum value resulting in an indicator whose high values corresponded to areas of low habitat risk.

2.4. Representation of changes in ecosystem services supply

The impact of any particular development is typically assessed at the feature level, i.e. a particular design feature such as housing development, road infrastructure, land use patches, administrative or watershed boundaries, and others (Emmett et al., 2015; Grafius et al., 2016; Mouchet et al., 2017; Qiu and Turner, 2015), yet ecosystem service provision is often modelled on a spatial grid better representing the natural spatial variability of environmental processes. There is therefore a translational step required that transforms grid based data to a polygon or feature based assessment. When the purpose is to report the magnitude of the biophysical indicator of a particular ecosystem service, the typical approach is to calculate the mean or sum of values from the cells in a gridded map that are enclosed within the features of interest. When changes in the supply of ecosystem services, expressed as the ESR or MESSI indices, are to be summarized within any type of a spatial feature, there are two possible approaches that can be undertaken. One approach is to derive the ESR or MESSI indices for each cell of the gridded map first, which can be easily done with map algebra capability of any GIS software, and then calculate the mean within the spatial features of interest ('*calculate first then scale*' approach). Another approach is to calculate sums of biophysical indicators for all time steps within features first, and then use these values to calculate ESR and MESSI indices ('*scale first then calculate*' approach). Despite originating from the same set of biophysical indicators values, both approaches are capable of yielding contrasting results, and this study explores the consequences of using either on the final outcome of an ecosystem services assessment.

In this study, changes in ecosystem services supply between the baseline and subsequent future conditions were calculated for the green infrastructure classes – grassland, scrubland, and woodland – introduced to the landscape with the design of the planned road, the planned road itself, and the entire study area encompassing the land approximately 500 m around the road axis. The assessment was also performed for land use patches reflecting the land use transition between baseline and future land use. All GIS operations required for the analysis were performed in ArcGIS 10.3 (ESRI, 2011) environment. The map of land use transitions was formed by the intersection of land use maps for the baseline and future conditions using the Intersect tool. The Raster Calculator tool was used to calculate the ESR and MESSI indices on grid-by-grid basis, whilst the Zonal Statistics as Table tool was used to calculate the sum of biophysical indicators of each ecosystem service ('*scale first then calculate*' approach) or the mean of ESR or MESSI ('*calculate first then scale*' approach) within the future land use classes for each time step. Thus obtained summaries where exported to a spreadsheet (Microsoft Excel 2010) whereby the ESR ratios were calculated and MESSI values averaged from the ESR ratios for '*scale first then calcu-*

late' approach, and radar plots depicting changes in the ecosystem services supply constructed.

3. Results and discussion

3.1. Consequences of using various approaches to up-scaling of ESR and MESSI values

The ESR and MESSI values obtained from gridded maps of modelled biophysical indicators of ecosystem services for each milestone year considered in this study were up-scaled to the extent of the study area using two approaches (Table 3). It is evident that the '*calculate first then scale*' approach returned more extreme ESR and corresponding MESSI values than the '*scale first then calculate*' approach. Although all ecosystem services assessed in this study were affected, sediment retention was characterized with particularly high ESR when it was calculated on the grid-by-grid basis first and subsequently averaged over the study area. This extreme result was caused by very large differences in modelled sediment retention values in individual pairs of grid cells in the maps for the baseline and future conditions. As opposed to upscaling of the modelled ecosystem services values prior to the calculation of the ESR, this caused very high ESR values assigned to multiple grid cells of the map and consequently yielded very high ESR values when averaged over the entire study area.

The differences between the magnitudes of ESR calculated with the two approaches can be explained by the non-linear character of the ESR values derived on the grid-by-grid cell basis, and the fact that this non-linearity was reduced by upscaling the modelled values to the entire study area in the '*scale first then calculate*' approach before calculation of the ESR. Non-linearity is common for natural phenomena (Corstanje et al., 2008; Corstanje and Lark, 2008), and may cause particularly large discrepancies in the ESR values calculated with the two differing approaches for ecosystems services showing high heterogeneity of modelled biophysical indicator values across space (water yield, and nitrogen, phosphorus and sediment retention). As seen in our example, these discrepancies may be substantial, and not only drive the magnitude of MESSI, but also may lead to contrasting conclusions with regards to changes in ecosystem services supply due to land use change such as in the case of water yield and nutrient retention services presented here. When used in spatial planning context, this may consequently lead to decisions that fail to optimize the supply of ecosystem services as a result of a development. When the purpose of the assessment is to estimate changes in ecosystem services over land use types or areas of interest whose size exceeds the grid size of the modelling outputs, we recommend using the '*scale first then calculate*' approach for calculation of the ESR as it is less sensitive to the non-linear effects in the data and produces more realistic values. The '*calculate first*' approach, however, could potentially be used to identify patterns of changes in ecosystem services within individual land use patches from gridded maps representing ESR or MESSI and determine areas with particularly high gains or losses of services unless the patterns depicted are very complex.

The differences resulting from the approach to calculation of the ESR can be propagated into maps showing overall changes in ecosystem services supply across milestone years. Here we present three sets of maps of a fragment of the study area for which the MESSI index was derived from ESR values calculated for all ecosystem services assessed in this study using three different approaches (Fig. 2). The first set of maps depicts MESSI calculated for each grid cell of the map separately, resulting in a highly complex picture of overall changes in the supply of ecosystem services taking into account changes between the neighboring grid cells in the map. This grid-cell to grid-cell heterogeneity is reduced by upscaling of the gridded maps to future land use classes, which was done with the '*calculate first*' and the '*scale first*' approaches. Maps produced with either approach vary in terms of the depicted levels of change in ecosystem services supply for reasons discussed earlier in this section. These maps demonstrate an advantage of representing the outcomes of an ecosystem services assessment within spatial summary units that are at a higher resolution than the study area but lower than the resolution of the gridded ecosystem services model outputs. This reduces the complexity of the gridded maps to spatially meaningful features and allows for the identification of portions of the study area that are particularly beneficial or detrimental in terms of ecosystem services supply, even if the overall assessment has not highlighted any issues. This could be of particular importance at a planning stage of a development where changes to the design could be applied and investment in green infrastructure made to mitigate negative effects of the development.

3.2. Implications of the new highway design on ecosystem services supply

The overall effect of the new major road development with associated green infrastructure on the supply of ecosystem services assessed in this study is projected to be neutral or moderately positive across all milestone years as seen from the MESSI metric calculated for the entire study area (Table 3, '*scale first then calculate*' approach). The supply of individual ecosystem services, however, is likely to vary depending on the type of service and time of assessment. In the first milestone year, the supply of all ecosystem services but sediment retention and water yield is likely to decline, with nitrogen retention maintaining a stable state. In the second milestone year, nitrogen retention is likely to achieve a positive state, and the amount of sequestered carbon is likely to achieve a state comparable to the baseline, whilst sediment retention and water yield supply are projected to continue at an increased level. In the final milestone year, all ecosystem services apart from arable production and avoided habitat risk are likely to be in a positive state of change.

Our results suggest that it is important to consider the time lag between the occurrence of land use change and the time of ecosystem services assessment as four ecosystem services assessed here exhibited differences in magnitude of change with time. In particular, carbon sequestration and nutrient retention services were affected by a reduction of supply in the first milestone year followed by a gradual increase in the supply over time as compared to the baseline. For carbon sequestration this is likely caused by disturbance of soil carbon and removal of biomass that was there before planting of new vegetation (Thuille and Schulze, 2006), followed by gradual accumulation of carbon in the new land use. In the case of nutrient retention, which was modelled without taking into account any bio-chemical interactions other than vegetation filtration capacity and nutrient loadings in each land use that were kept constant for both nutrients across the milestone years, differences in magnitudes are likely due to different magnitudes in nutrient loadings in the initial and subsequent land use as well as the amount of available water in the landscape. Water yield displayed the highest increase as compared to the baseline with a decreasing trend of changes induced by the decreasing availability of rainwater run-off due to the interactions between the introduced sealed surface of the road, rainfall levels, and increased evapotranspiration from the area. The remaining ecosystem services assessed here did not display varied levels of change across time. Arable production within the boundaries of the study area was projected to be in a stable declined state resulting from less land being available for agricultural use. Avoided habitat risk was modelled under the assumption of constant risk imposed by the road onto the existing and new habitats and therefore did not

Table 3
ESR and MESSI values calculated for the entire study area using the '*calculate first then scale*' approach where the ESR index was calculated on grid cell basis and then summarized over the study area, and the '*scale first and then calculate*' approach where the biophysical values of ecosystem services indicators were first summarized within the study area followed by calculation of the ESR. ES – ecosystem service, AP – arable production, AH – avoided habitat risk, CS – carbon sequestration, NR – nitrogen retention, PR – phosphorus retention, SR – sediment retention, WY – water yield.

ES	'Calculate first' approach			'Scale first' approach		
	Milestone year					
	2017	2032	2077	2017	2032	2077
AH	0.51	0.51	0.51	0.84	0.84	0.84
AP	0.15	0.15	0.15	0.23	0.23	0.23
CS	0.90	0.99	1.23	0.89	0.98	1.22
NR	8.23	9.47	9.60	0.99	1.08	1.15
PR	6.70	7.76	8.27	0.84	0.92	1.00
SR	198,232.56	197,908.72	197,908.72	1.68	1.67	1.67
WY	152.97	166.76	176.36	2.62	2.18	1.94
MESSI	28,343.15	28,299.19	28,300.69	1.16	1.13	1.15

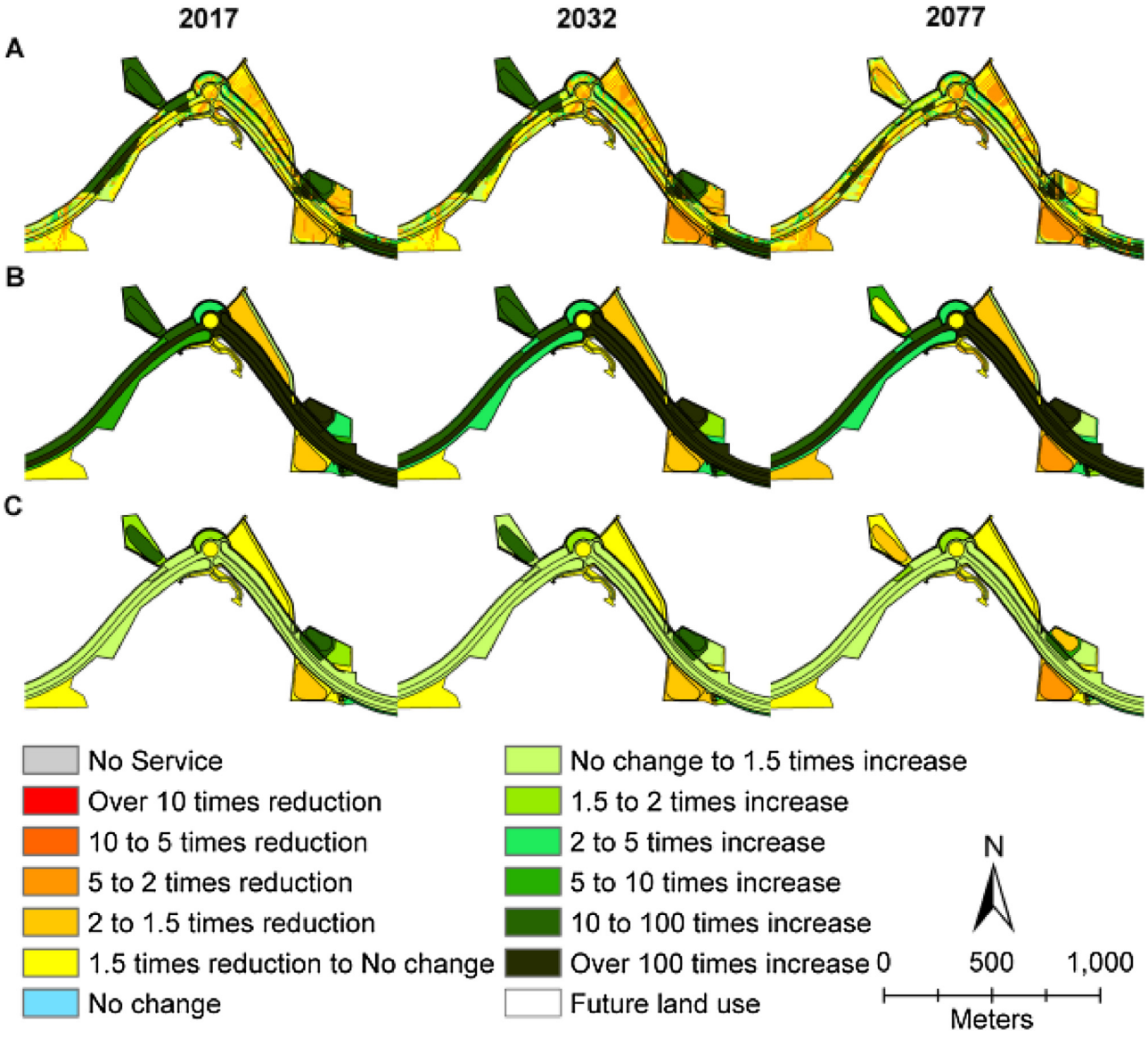

Fig. 2. Gridded and scaled maps of the Modified Ecosystem Services Status Index (MESSI) obtained for a fragment of the planned road across all milestone years: A – MESSI calculated individually for each grid cell of the map, B – '*calculate first then scale*' approach, C – '*scale first then calculate*' approach.

show variation in the following milestone years. Sediment retention in the first milestone year was only slightly higher than in the following years, attributed to differing values of the rainfall erosion index of the revised USLE soil erosion model, resulting from diminishing values of rainfall projected for the study area over time.

Consideration of individual land use classes provides further insights as to the likely effects of particular components of the

Table 4
ESR and MESSI values calculated with the '*scale first then calculate*' approach for the planned road and three green infrastructure land use classes introduced with the road for the three milestone years assessed for the future conditions.

LU	Year	ESR							MESSI
		AH	AP	CS	NR	PR	SR	WY	
Road	2017	0.00	0.00	0.51	0.61	0.53	1.24	4.27	1.02
	2032	0.00	0.00	0.51	0.70	0.61	1.24	4.17	1.03
	2077	0.00	0.00	0.51	0.79	0.69	1.24	4.14	1.05
Grassland	2017	0.85	0.00	0.97	1.24	1.03	2.22	2.86	1.31
	2032	0.85	0.00	1.01	1.37	1.13	2.22	1.78	1.19
	2077	0.85	0.00	1.10	1.46	1.23	2.22	1.15	1.15
Scrubland	2017	0.81	0.00	0.90	1.65	0.92	1.96	2.40	1.23
	2032	0.81	0.00	1.02	1.89	1.06	1.96	1.64	1.20
	2077	0.81	0.00	1.13	1.93	1.10	1.96	1.19	1.16
Woodland	2017	0.80	0.00	0.97	1.15	0.89	2.48	0.51	0.97
	2032	0.80	0.00	1.38	1.28	0.98	2.48	0.36	1.04
	2077	0.80	0.00	2.47	1.38	1.07	2.48	0.28	1.21

LU – Land use, AH – avoided habitat risk, AP – arable production, CS – carbon sequestration, NR – nitrogen retention, PR – phosphorus retention, SR – sediment retention, WY – water yield.

development design on the future supply of ecosystem services (Table 4). The road itself is expected to be neutral in terms of the average change in provision of ecosystem services across all development milestone years (MESSI ranging between 1.02 and 1.05). Further enquiry into the change in individual ecosystem services indicated by the ESR values, however, reveals large losses of the food and habitat provisioning services, as well as carbon storage and nutrient retention imposed by the new road. The only services that are likely to benefit from the land use change are sediment retention and water yield, the latter driving the overall non-negative outcome of the assessment due to a large magnitude of change (four times the baseline). Before this additional amount of water could be used for hydropower production suggested by the InVEST model, it would have to be directed to watercourses. Road runoff is often contaminated by heavy metals, deicing salts, organic compounds, and other pollutants (Berhanu Desta et al., 2007; Rivett et al., 2016), which may affect aquatic habitats (Corsi et al., 2010; Maltby et al., 1995) and therefore should be treated before releasing into water courses. In our case study, the excess run-off is collected in purposefully designed vegetated lagoons that have a potential for treatment of road run-off pollutants (Roinas et al., 2014) as well as slowing down of water flow which can be retained and contribute to maintaining water balance within the area. In terms of the green infrastructure land use classes, their impact is likely to vary with the land use type and is similar between grassland and scrubland being distinct to the one of woodland. Both grassland and scrubland offer a large improvement in sediment and nitrogen retention, with a positive long-term effect on remaining services apart from arable production and habitat provision. The distinct behavior of ecosystem services change under woodland is characterized by a reduction in water yield across all milestone years and a large positive change in carbon sequestration, especially in the final year of the assessment. Woodland appears to be most beneficial for the sediment retention service, however, offers lower benefits for nutrient retention, particularly for phosphorus.

The different behaviors of various land use classes in terms of changes in the supply of ecosystem services is consistent with the idea of ecosystem services bundles defined as the sets of services that appear together repeatedly (Raudsepp-Hearne et al., 2010) associated with particular landscape features (Bieling and Plieninger, 2013), except that in our case we focus on groups of ecosystem services with similar direction and magnitude of change induced by new land use. Their identification is facilitated by visualization of the data with radar plots (Fig. 3) that allow for "at-a-glance" assessment of positive or negative changes indicated by the positioning of the plotted lines above or below the marked baseline as well as similarities and differences between changes in ecosystem services supply across various final land uses and milestone years expressed as the shape of the plotted lines. The plots confirm that grassland and scrubland are likely to induce similar patterns of change, especially in the final milestone year, and that woodland and the new road are likely to have a different impact on ecosystem services supply.

3.3. Implications of the land use transitions in ecosystem services assessments

As so far we discussed results based on summaries of modelled biophysical indicators of ecosystem services within all patches of the same land use type regardless of their location within the study area. In any study area it is possible that the baseline conditions for different patches of the same future land use type are different, and this may change the amplitude of the improvement or deterioration in the supply of a given service across space not accounted for in the summaries considering future land use only. For example, carbon sequestration benefits in land use conversion from no-till agriculture to grassland may appear smaller than in the case when conventional arable fields constitute the initial land use. Conversely, introduction of any vegetation to derelict land is likely to improve the habitat suitability for general biodiversity to a greater degree than introduction of the same land use on land exhibiting ecological function in the baseline state. This means that the choice of the future land use could be locally customized to meet the demands for ecosystem services supply of the area, i.e. to maximize the benefits and minimize the losses; or achieve a comparable level of ecosystem services change by locally introducing a land use that requires fewer resources to implement and maintain than the land use indicated in the generic assessment. This approach potentially calls for implementation of detailed data about the study area to the assessment, including land management methods, influence of which on ecosystem services is becoming evident in literature (Ruhl, 2016). These could be collected as part of surveys conducted for the environmental impact assessment or obtained from stakeholder participatory sessions. Since the InVEST models work on user-customized data tables, such information could be relatively easily incorporated into the analyses (Tallis and Polasky, 2009). Alternatively, when the study

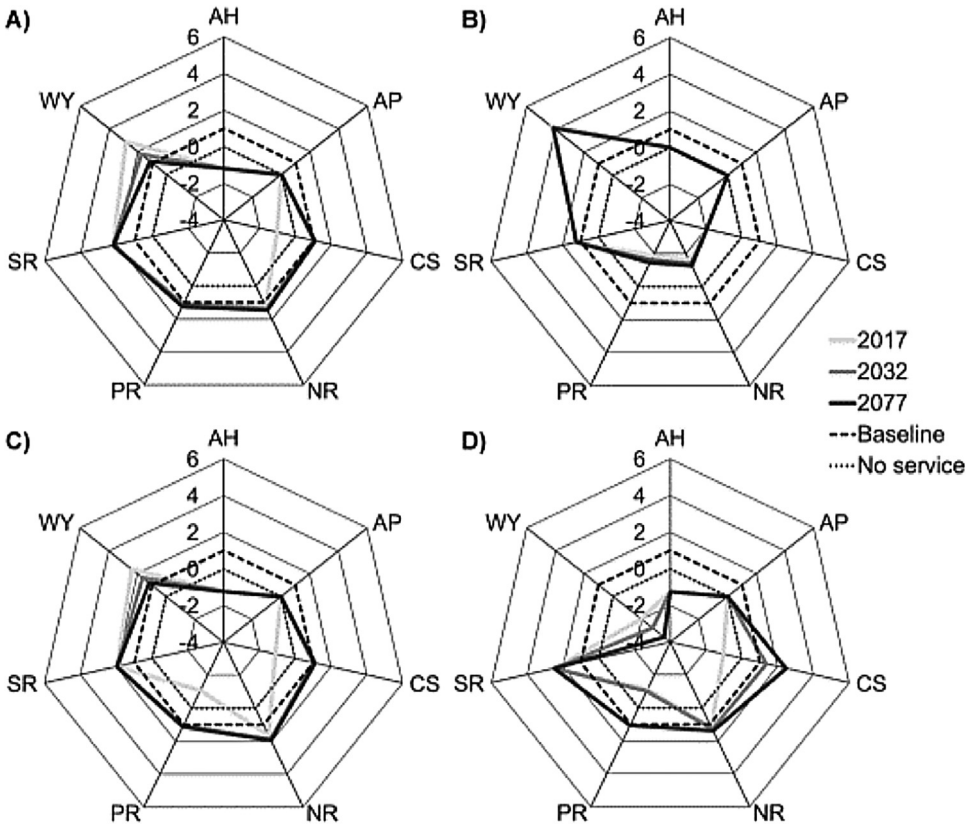

Fig. 3. Radar plots depicting changes in ecosystem services supply in green infrastructure land use classes introduced with the planned road, as well as the road itself, calculated between the three milestone years and the baseline, using the '*scale first then calculate*' approach. Refer to Table 4 for explanations to ecosystem services abbreviations. A – Grassland, B – Major road, C – Scrubland, D – Woodland.

area is large and the information on ecosystem services supply is inferred from generic land use maps, the land use transitions, i.e. the intersection of baseline and future land uses, could be taken into consideration in the customization process. This could help inform the best choice of the future land use given the initial state of ecosystem services supply inferred from the initial land use.

In our case study, consideration of the land use transitions reveals further differences in the changes in the ecosystem services supply from the final land uses. For example, considering changes between the baseline and milestone year 2032 (Fig. 4), it appears that grassland and scrubland are likely to produce different effects on water yield when arable land is the initial land use, on water yield and phosphorus retention in land use transition from broadleaved forest, and only exhibit similar behavior in land use transition from pastures. Conversely, woodland, previously deemed to reduce water yield, is only likely to do so if the initial land use is pasture.

Incorporation of land use transitions into ecosystem services assessment would be particularly valuable at the scoping stage when the location and configuration of a development is being decided. The process could be facilitated by purpose-built tools similar to existing tools (Gonzalez-Redin et al., 2015; McVittie et al., 2015) that use the flexibility and decision support capacity of Bayesian Belief Networks, making it possible to incorporate economic valuation and analysis of the demand for ecosystem services into the assessment. That would allow for gradation of importance of each ecosystem service and help optimize, i.e. maximize gains and minimize losses, the future land use choices to make decisions that take into account the needs of both the environment and the society.

4. Conclusions

We presented a study of changes in the supply of seven ecosystem services belonging to the provisioning, regulating and supporting categories quantified from the outcomes of spatially explicit ecosystem services models with the use of two simple metrics. We discussed possible methods for the upscaling of spatial data into summary units used in planning decisions and their consequences to the validity of the assessment. We found that the changes in ecosystem services supply due to the introduction of a major road with associated green infrastructure were dominated by the loss of the provisioning service of agricultural production in exchange for the supply of other ecosystem services owing to the incorporation of woodland, scrubland and grassland into the design of the development. These changes varied not only with the introduced land use and ecosystem service type, but also depended on the initial land use and the temporal dimension defined by the time lag between the intervention to the landscape and time of assessment.

When compared to the traditional environmental impact assessment, the ecosystem services approach conducted in this study offered a possibility of appraisal of both the positive and negative impacts of the development as opposed to making inventories of risks and proposed mitigation options. Contrary to the EIA, it generated a low-volume and compact report, which can be benefit retrieval of information for the decision making process. The relative ease of application makes the assessment particularly suitable at the scoping stage of a development leading to a design that optimizes the flows of natural capital in the affected area. This approach could be further facilitated by development of decision support tools that could be customized with the data routinely collected

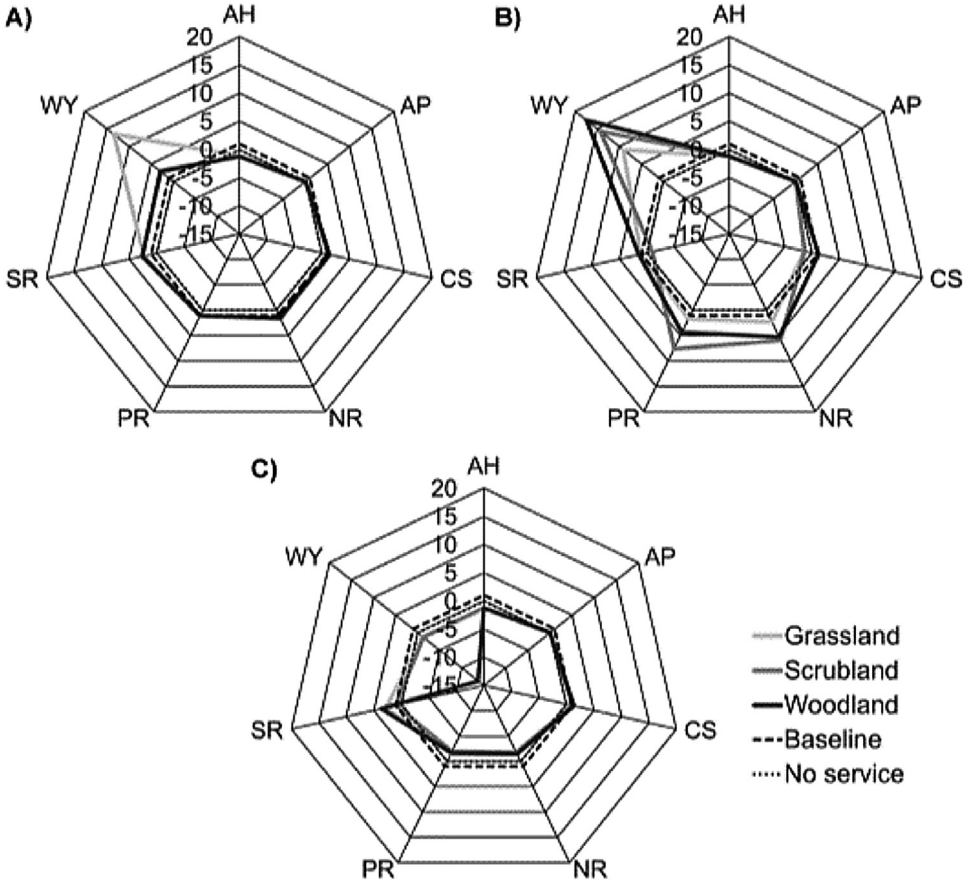

Fig. 4. Ecosystem services ratio for all ecosystem services assessed in this study in selected land use transitions from baseline to future land use for the milestone year 2032 and selected initial land uses. A – land use change from arable, B – land use change from broadleaved forest, C – land use change from pastures.

during the process of impact assessment, including on-the-ground surveys and stakeholder participation sessions, as well as economic valuation and demand for services, and consequently provide an accurate picture of changes to the natural capital, economy and society on which planning decisions can be based.

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References

Baker, J., Sheate, W.R., Phillips, P., Eales, R., 2013. Ecosystem services in environmental assessment – help or hindrance? Environ. Impact Assess. Rev. 40, 3–13, http://dx.doi.org/10.1016/j.eiar.2012.11.004.

Berhanu Desta, M., Bruen, M., Higgins, N., Johnston, P., 2007. Highway runoff quality in Ireland. J. Environ. Monit. 9, http://dx.doi.org/10.1039/b702327h.

Bieling, C., Plieninger, T., 2013. Recording Manifestations of Cultural Ecosystem Services in the Landscape. Landsc. Res. 38, 649–667, http://dx.doi.org/10.1080/01426397.2012.691469.

Cashmore, M., Bond, A., Cobb, D., 2007. The contribution of environmental assessment to sustainable development: toward a richer empirical understanding. Environ. Manage. 40, 516–530, http://dx.doi.org/10.1007/s00267-006-0234-6.

Corsi, S.R., Graczyk, D.J., Geis, S.W., Booth, N.L., Richards, K.D., 2010. A fresh look at road salt: aquatic toxicity and water-quality impacts on local, regional, and national scales. Environ. Sci. Technol. 44, 7376–7382, http://dx.doi.org/10.1021/es101333u.

Corstanje, R., Lark, R.M., 2008. On effective linearity of soil process models. Eur. J. Soil Sci. 59, 990–999, http://dx.doi.org/10.1111/j.1365-2389.2008.01046.x.

Corstanje, R., Grunwald, S., Lark, R.M., 2008. Inferences from fluctuations in the local variogram about the assumption of stationarity in the variance. Geoderma 143, 123–132, http://dx.doi.org/10.1016/j.geoderma.2007.10.021.

ESRI, 2011. ArcGIS Desktop: Release 10.

Emmett, B.A., Cooper, D., Smart, S., Jackson, B., Thomas, A., Cosby, B., Evans, C., Glanville, H., McDonald, J.E., Malham, S.K., Marshall, M., Jarvis, S., Rajko-Nenow, P., Webb, G.P., Ward, S., Rowe, E., Jones, L., Vanbergen, A.J., Keith, A., Carter, H., Pereira, M.G., Hughes, S., Lebron, I., Wade, A., Jones, D.L., 2015. Spatial patterns and environmental constraints on ecosystem services at a catchment scale. Sci. Total Environ., http://dx.doi.org/10.1016/j.scitotenv.2016.04.004.

Glasson, J., Therivel, R., Chadwick, A., 2012. Introduction to environmental impact assessment. In: The Natural and Built Environment Series, 4th ed. Taylor & Francis Ltd., London, United Kingdom.

Gonzalez-Redin, J., Luque, S., Poggio, L., Smith, R., Gimona, A., 2015. Spatial Bayesian belief networks as a planning decision tool for mapping ecosystem services trade-offs on forested landscapes. Environ. Res. 144, 15–26, http://dx.doi.org/10.1016/j.envres.2015.11.009.

Grafius, D.R., Corstanje, R., Warren, P.H., Evans, K.L., Hancock, S., Harris, J.A., 2016. The impact of land use/land cover scale on modelling urban ecosystem services. Landsc. Ecol. 31, http://dx.doi.org/10.1007/s10980-015-0337-7.

Gratani, L., Varone, L., 2013. Carbon sequestration and noise attenuation provided by hedges in Rome: the contribution of hedge traits in decreasing pollution levels. Atmos. Pollut. Res. 4, 315–322, http://dx.doi.org/10.5094/APR.2013.035.

Jaggard, K.W., Qi, A., Ober, E.S., 2010. Possible changes to arable crop yields by 2050. Philos. Trans. R. Soc. Lond. B: Biol. Sci. 365, 2835–2851, http://dx.doi.org/10.1098/rstb.2010.0153.

Jay, S., Jones, C., Slinn, P., Wood, C., 2007. Environmental impact assessment: retrospect and prospect. Environ. Impact Assess. Rev. 27, 287–300, http://dx.doi.org/10.1016/j.eiar.2006.12.001.

Leh, M.D.K., Matlock, M.D., Cummings, E.C., Nalley, L.L., 2013. Quantifying and mapping multiple ecosystem services change in West Africa. Agric. Ecosyst. Environ. 165, 6–18, http://dx.doi.org/10.1016/j.agee.2012.12.001.

Maltby, L., Forrow, D.M., Boxall, A.B.A., Calow, P., Betton, C.I., 1995. The effects of motorway runoff on freshwater ecosystems: 1. Field study. Environ. Toxicol. Chem. 14, http://dx.doi.org/10.1002/etc.5620140620.

McVittie, A., Norton, L., Martin-Ortega, J., Siameti, I., Glenk, K., Aalders, I., 2015. Operationalizing an ecosystem services-based approach using Bayesian Belief

Networks: an application to riparian buffer strips. Ecol. Econ. 110, 15–27, http://dx.doi.org/10.1016/j.ecolecon.2014.12.004.
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Synthesis, Washington, DC.
Mouchet, M.A., Paracchini, M.L., Schulp, C.J.E., Stürck, J., Verkerk, P.J., Verburg, P.H., Lavorel, S., 2017. Bundles of ecosystem (dis)services and multifunctionality across European landscapes. Ecol. Indic. 73, http://dx.doi.org/10.1016/j.ecolind.2016.09.026.
Opdam, P., Coninx, I., Dewulf, A., Steingröver, E., Vos, C., van der Wal, M., 2015. Framing ecosystem services: affecting behaviour of actors in collaborative landscape planning? Land Use Policy 46, 223–231, http://dx.doi.org/10.1016/j.landusepol.2015.02.008.
Pugh, T.A.M., Mackenzie, A.R., Whyatt, J.D., Hewitt, C.N., 2012. Effectiveness of green infrastructure for improvement of air quality in urban street canyons. Environ. Sci. Technol. 46, 7692–7699, http://dx.doi.org/10.1021/es300826w.
Qiu, J., Turner, M.G., 2015. Importance of landscape heterogeneity in sustaining hydrologic ecosystem services in an agricultural watershed. Ecosphere 6, http://dx.doi.org/10.1890/es15-00312.1.
Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M., 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc. Natl. Acad. Sci. U. S. A. 107, 5242–5247, http://dx.doi.org/10.1073/pnas.0907284107.
Rivett, M.O., Cuthbert, M.O., Gamble, R., Connon, L.E., Pearson, A., Shepley, M.G., Davis, J., 2016. Highway deicing salt dynamic runoff to surface water and subsequent infiltration to groundwater during severe UK winters. Sci. Total Environ. 565, http://dx.doi.org/10.1016/j.scitotenv.2016.04.095.
Rodríguez, J.P., Beard, T.D., Bennett, E.M., Cumming, G.S., Cork, S.J., Agard, J., Dobson, A.P., Peterson, G.D., 2006. Trade-offs across space, time, and ecosystem services. Ecol. Soc. 11, http://dx.doi.org/10.2307/2390206.
Roinas, G., Tsavdaris, A., Williams, J.B., Mant, C., 2014. Fate and behavior of pollutants in a vegetated pond system for road runoff. Clean Soil Air Water 42, http://dx.doi.org/10.1002/clen.201300159.
Ruhl, J.B., 2016. Adaptive management of ecosystem services across different land use regimes. J. Environ. Manage. 183, http://dx.doi.org/10.1016/j.jenvman.2016.07.066.
Silli, V., Salvatori, E., Manes, F., 2015. Removal of airborne particulate matter by vegetation in an urban park in the city of Rome (Italy): an ecosystem services perspective. Ann. di Bot. 5, 53–62, http://dx.doi.org/10.4462/annbotrm-13077.
Tallis, H., Polasky, S., 2009. Mapping and valuing ecosystem services as an approach for conservation and natural-resource management. Ann. N. Y. Acad. Sci. 1162, 265–283, http://dx.doi.org/10.1111/j.1749-6632.2009.04152.x.
Tallis, H., Ricketts, T., Guerry, A., Wood, S., Sharp, R., Nelson, E., Ennaanay, D., Wolny, S., Olwero, N., Vigerstol, K., Pennington, D., Mendoza, G., Aukema, J., Foster, J., Forrest, J., Cameron, D., Arkema, K., Lonsdorf, E., Kennedy, C., Verutes, G., Kim, C., Guannel, G., Papenfus, M., Toft, J., Marsik, M., Bernhardt, J., Griffin, R., Glowinski, K., Chaumont, N., Perelman, A., Lacayo, M., 2013. INVEST 2.5.6 User's Guide. The Natural Capital Project.
The Norfolk County Council, 2014. Norwich Northern Distributor Road (A1067 to A47(T)) Order: Environmental Statement.
Thuille, A., Schulze, E.-D., 2006. Carbon dynamics in successional and afforested spruce stands in Thuringia and the Alps. Global Change Biol. 12, 325–342, http://dx.doi.org/10.1111/j.1365-2486.2005.01078.x.
UNCED, 1992. Rio declaration on environment and development. Rio de Janeiro In: The United Nations Conference on Environment and Development, vol. 1 (No. A/Conf. 151/26).
Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall erosion losses: a guide to conservation planning. In: Agriculture Handbook No. 537. USDA/Science and Education Administration, US. Govt. Printing Office, Washington, DC.

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Operationalizing the ecosystems approach: assessing the environmental impact of major infrastructure development

Zawadzka, Joanna Ewa; Corstanje, Ronald; Fookes, Jacqueline; Nichols, Jonathan; Harris, Jim A.

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