Appendix 3. Monetary Quantification and valuation methods

WATER RETENTION

Quantification: Water retention occurs at sites that are at least temporarily waterlogged. Many of these waterlogged sites have been drained for agriculture, housing etc… We quantify the retention as the mean level of water saturation in the topsoil (% waterlogged up to 1 meter depth). In combination with the soil porosity, we can express this as a retention volume (m^3) per area unit. Potential water retention is derived from a) Mean historical high water levels, interpolated from soil data (indication of the depth of oxidation- reduction fronts); b) information on infiltration-seepage patterns at multiple scales, which is derived from a multi-scale topographic position index on a high resolution DEM (Jenness 2006, De Reu et al. 2013). Actual retention is limited by drainage intensity, which is derived from land-use intensity (desired drainage depth) and drainage network density (distance decay principle).

Valuation: Valuation of additional water retention was explored from two viewpoints. The first is the substitution cost: In times of scarcity, water is purchased in the Walloon region. A recent benchmark study of the drinking water companies revealed the average additional costs of purchasing treated water versus own production at the level of the Flemish Region. This difference in costs is approximately 0.2 €/m^3. The second method used is based on the groundwater water abstraction tax. This tax is 0.075 €/m^3 and can be seen as a compensation for the environmental and resource costs as formulated within the Water Framework Directive. This is the existing effective contribution from water companies and should be regarded as the absolute bottom threshold.

AGRICULTURAL PRODUCTION

Quantification: Because of the nature of the primary data, quantification is done directly in €/ha*yr. Typical agricultural net revenues per crop type are derived from sample data on profits and loss accounts at the farm level. These values are then used in combination with the parcel level crop registration data of 2010 and crop specific soil suitability maps to account for spatial variations in crop specific productivity. The profits and loss accounts reflect the state of revenues and costs for particular agricultural sectors. The net revenue is the difference between the total revenue from agricultural production (excluding subsidies) minus the operational costs. For the year 2010 this was derived from detailed data from a random check of 749 particular farms (Van Broekhoven E. 2010). Because of the variability between years we used data from 2008, 2009 and 2010 to estimate the values per crop type. For fodder crop types, an alternative method is used by the agricultural administration (D'Hooghe 2012). In general, fodder crops are not sold on the market, but are used as fodder within the agricultural production chain. The net revenues from dairy and meat production are therefore distributed among the fodder production parcels at the farm level to estimate a revenue factor for fodder crops. Based on this data, a P25, P50 and P75 revenue value (per ha) was derived for the most important crop types (e.g. for corn P25=€ 1.245, P50=€1.580, P75=€1.818).

The soil suitability maps for agriculture and horticulture are based on the digital soil map of Belgium (Dudal et al. 2005) A suitability class groups the soil types that can provide a comparable production for several crop types when uniform cultivation and fertilization practices are applied. For each crop type, a 5 class ranking is provided, where class 1 is very suitable and class 5 is unsuitable (Bollen

2012). The classes 1-2 are associated with the P75 values of the crop revenue values, the classes 3-4 are associated with the P50 values and class 5 is associated with the P25 values.

WOOD PRODUCTION

Quantification: Wood production depends on soil characteristics and applied harvest regime. Species specific potential produced wood volumes can be found in table A.1, where differentiation is made according to the soil suitability.

Table A1: Overview of the relationships between soil suitability and the maximal mean growth of stemwood (m^3/ha*yr).

Soil suitability	Tree species							
	Fagus sylvatica	Quercus (Robur, Rubra)	Populus	Larix decidua	Pinus sylvestris	Pinus Nigra	Picea abies	Pseudotsuga menziesii
4	4.0	3.0	9.0	6.0	4.0	6.0	6.0	8.0
3	6.7	5.0	11.0	8.7	6.0	9.3	9.0	10.7
2	9.3	7.0	13.0	11.3	8.0	12.7	13.0	13.3
1	12.0	9.0	15.0	14.0	10.0	16.0	16	16.0

Depending on management and ownership structure (private, public) a harvest factor is applied that estimates the proportion of the annual maximal mean growth that is harvested annually. The harvested volumes are available from recent data (2009-2012) on timber selling from public (state owned) forests and from forest owner cooperatives (privately owned, but the management is state coordinated). This data base has about 80.000 records of sold volumes per tree species and circumferences. For state owned forests, the harvest factor is 0.54. Privately owned forests are often unmanaged and have a lower (0.15) harvest factor. For private forests, there is an unknown fraction of harvest for private use and informal markets (especially for fire wood).

Valuation: Valuation of wood production has been done on the basis of annual m³ harvest per year and per tree species. The value for each species was based on the database of actual selling prices in the state-owned forests for the years 2009-2012. Although the records refer to tree species, volumes and associated circumferences, the selling prices often refer to a combination of several records sold as one single lot (in average 18 records/lot). Statistical analysis (SPSS 20.0) was used to reveal a selling price (€/m³) per species and circumference class (Table A.2). The average weighted selling price for all species and circumferences was estimated at 32.43 €/m³. Trees are sold as standing timber and prices are therefore considered as net added value of timber production.

Table A2: Overview of timber values (€ per m³) sold as standing timber per circumference class for most important commercial tree species.

Circumference (cm)	Tree species								
	Fagus sylvatica	Quercus Robur/petrea	Quercus Rubra	Populus	Larix decidua	Pinus sylvestris	Pinus Nigra	Picea abies	Pseudotsuga menziesii
100–119	30.6	27.0	27.1	27.1	25.6	26.9	28.7	24.3	28.5
120-149	33.7	30.7	30.6	30.6	29.4	27.7	29.1	24.9	31.7
150-179	39.9	41.4	36.7	36.7	33.7	29.4	30.2	26.0	33.3
180-199	43.6	45.8	38.4	38.4	37.0	27.4	33.6	28.8	34.9
200–219	48.1	48.3	39.1	39.1	41.6	32.3	32.0	25.4	37.8
220-249	48.8	50.2	43.0	43.0	45.6	-	33.5	-	35.1
>250	50.4	52.8	43.1	43.1	- $	-	-	-	32.4
Average £	39.47	35.99	33.64	36.42	29.95	27.55	29.54	24.97	31.95

$: n<10 unrealiable parameter estimation;

£: weighted mean, based on number of observations per circumference

CARBON SEQUESTRATION IN SOILS

Quantification: Soils under unmanaged, natural vegetation types typically have larger carbon stocks than managed vegetation types. Also soil hydrology plays a crucial role in the creation of soil organic carbon (SOC) stocks. Soil organic carbon is especially high for forests and/or hydric soils. The potential equilibrium state for soil organic carbon stocks can be calculated using the regression formula by Meersmans et. al. (2008), which includes parameters like water retention, soil texture and vegetation type. Changes in land-use typically affect both vegetation and/or drainage (ES water retention), which leads to a new potential equilibrium state for SOC stocks. Recent research by Dr. De Vos (2009) has revealed that this function systematically underestimated SOC-stocks in forest soils with 32 %. This correction factor to the regression formula of Meersmans is applied to all forests. Peatlands, wetlands and freshwater ecosystems can sequester higher carbon stocks than terrestrial ecosystems. Potential (maximal) stocks are very much dependent on hydrological regimes and how mature these ecosystems are. Depending on the hydrological regime, newly created wetlands sequester 2.5-3.5 ton C/ha*yr in the first 100 years. Older wetland systems often do not sequester much additional carbon, especially when they are not under permanent hydric conditions. On the other hand, pulsed hydrological conditions emit less methane.

Valuation: Stocks are difficult to consider in valuation exercises. Here we calculated a virtual scenario of changes in carbon stocks due to changes in land use (habitat types) and associated changes in water retention. The difference in SOC stocks would be gradually built or released at a rate of 2.5 % loss per year. The valuation method is identical to the valuation of carbon sequestration in biomass.

AVOIDED NITRATE LEACHING

Quantification: It can be debated if the cessation of fertilizer use can be categorized as an ES. It is imperative to include this since landscape level nutrient leaching is an important parameter for the ES “nutrient removal by denitrification”. Infiltration on fertilized agricultural land results in nitrate leaching to groundwater and eventually surface water. Important variables are the specific combinations of soil texture, crop type, agricultural fertilizer application (kg N/ha) and atmospheric N-deposition (kg N/ha). Long-term data on autumn and spring nitrate residues in agricultural soils were available from the Flemish Land Agency (Geypens et al. 2005). The difference between fall and spring residue is assumed to be leached out by winter precipitation. Atmospheric nitrogen deposition data were provided by the Flemish Environment Agency (FEA 2011). We assumed that nutrient leaching also occurs on non-agricultural land with high deposition rates. Although declining, these values are still relatively high (Staelens et al. 2012). From the data on nutrient leaching from agricultural land we know these values range between 7 % and 33 % of the nitrate application. We applied the same range of values (7 - 33 %) on non-agricultural land with N-deposition and varied the range of values accordingly to the natural sensitivity for nitrate leaching (soil texture).

Table A3: parameter values for maximal fertilizer application, fall nitrate residues in soils and winter nitrate leaching in function of cultivation and soil texture.

Cultivation	Texture	Max. N-application kg N/ha	N-residue fall	Relative residue (%)	N-leaching	Relative leaching (%)
pasture	sand	350	60	17%	32	9%
pasture	loam	370	67	18%	26	7%
pasture	clay	380	73	19%	23	6%
beet (fodder)	sand	305	49	16%	30	10%
beet (fodder)	loam	330	55	17%	24	7%
beet (fodder)	clay	330	60	18%	21	6%
maize	sand	205	86	42%	57	28%
maize	loam	220	96	44%	40	18%
maize	clay	220	105	48%	41	19%
barley and other cereals	sand	200	69	35%	42	21%
barley and other cereals	loam	215	77	36%	33	15%
barley and other cereals	clay	215	84	39%	30	14%
wheat and tricitale	sand	250	80	32%	42	17%
wheat and tricitale	loam	264	89	34%	33	13%
wheat and tricitale	clay	265	98	37%	29	11%
crops with low N-demand	sand	165	69	42%	42	25%
crops with low N-demand	loam	175	76	44%	33	19%
crops with low N-demand	clay	175	84	48%	29	17%
other	sand	50	9	18%	5	10%
other	loam	50	10	20%	4	8%
other	clay	50	11	22%	4	8%
Vegetables Group II	sand	180	86	48%	53	29%
Vegetables Group II	loam	180	96	53%	41	23%
Vegetables Group II	clay	180	105	58%	37	21%
potatoes	sand	280	111	40%	68	24%
potatoes	loam	280	124	44%	53	19%
potatoes	clay	280	136	49%	48	17%
Sugar beet	sand	205	54	26%	33	16%
Sugar beet	loam	220	60	27%	26	12%
Sugar beet	clay	220	66	30%	23	10%
Vegetables Group III	sand	125	66	53%	40	32%
Vegetables Group III	loam	125	74	59%	32	26%
Vegetables Group III	clay	125	81	65%	28	22%
Vegetables Group I	sand	250	114	45%	69	28%
Vegetables Group I	loam	250	126	50%	54	22%
Vegetables Group I	clay	250	139	56%	49	20%
crops	sand	200	90	45%	55	28%
crops	loam	215	100	46%	43	20%
crops	clay	215	110	51%	38	18%
legumes (other than peas and beans)	sand	120	39	32%	24	20%
legumes (other than peas and beans)	loam	125	43	35%	19	15%
legumes (other than peas and beans)	clay	125	48	38%	17	14%

Valuation: see below section on nutrient removal by denitrification.

NUTRIENT REMOVAL BY DENITRIFICATION

Quantification: Under conditions of (temporal) waterlogging, bacterial processes enable to remove nitrogen from ground and surface water. The most important variables are the soil moisture, supply of nitrate, residence time and soil organic carbon. As a proxy for nitrate removal efficiency, we transform combinations of the mean highest (MHG) and mean lowest groundwater (MLG) levels to an estimated nitrate removal efficiency (% of available nitrate removed).

Table A4: Estimated removal efficiency (%) for combinations of mean highest and mean lowest groundwater levels (in cm below soil surface).

	MLG	>50	45	40	35	30	25	20	15	10	5-0
MHG											
>50		10	13	17	20	23	27	30	33	37	40
45			20	23	27	30	33	37	40	43	47
40				30	33	37	40	43	47	50	53
35					40	43	47	50	53	57	60
30						50	53	57	60	63	67
25							60	63	67	70	73
20								70	73	77	80
15									80	83	87
10										90	93
0-5											100

Nitrogen has many and complex pathways by which it is dispersed in the environment. For this study, we focus on the issue of excess nitrogen in groundwater and surface water. Nutrient leaching from agricultural land is one of the major pathways. Reduction of nitrate leaching has already been described in previous sections, but is an important variable for denitrification. For the current situation, the avoided nitrate leaching is zero. But the NCO's include both cessation of fertilizer application and cessation of drainage. Cessation of nitrate leaching implies a decrease of nitrate supply to the denitrification zones, which in their turn may have increased nitrate removal efficiency due to rewetting. The supply of nitrogen occurs through patterns of (local) infiltration (nitrate leaching) and seepage. Infiltration and seepage patterns are the result of processes that occur on a range of spatial scales. A topographic position index (TPI) is used to identify these patterns at multiple scales (Jenness 2006). This method has also been used in other studies for the Flemish Region and has proven its applicability (De Reu et al. 2013). We calculated the TPI at a range of spatial scales (radius: 250m – 2000m) to indicate these local infiltration-seepage patterns. The multi-scale TPI is then corrected for soil permeability to result in a seepage intensity map, indicating the water supply to a particular pixel (mm/day). The nitrate concentration of the supplied seepage water is calculated at the landscape level (2 km radius) by multiplying the annual nitrate leaching (kg N/ha) with the annual infiltration (m^3/ha). This allows us to calculate denitrification by multiplying the removal efficiency with the annual nitrate load for each pixel.

Valuation: The valuation is based on the marginal reduction cost for nitrate removal. The Environmental Costing Model for Flanders compares different (technical) measures on cost-efficiency (€/kg reduction) and the applicability of those measures. The cost of the most expensive measure, considered in policy approved measure programs, can be seen as the cost the society is willing to pay for a further reduction of nitrate levels in ground and surface water. For nitrate, the marginal reduction cost is 74 €/kg N. As a low estimate we apply 5 €/kg N, based on a literature review (Cools et al. 2011, Broekx et al. 2013a, Broekx et al. 2013b).

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