Economic benefits of reducing agricultural N losses to coastal waters for seaside recreation and real estate value in Denmark

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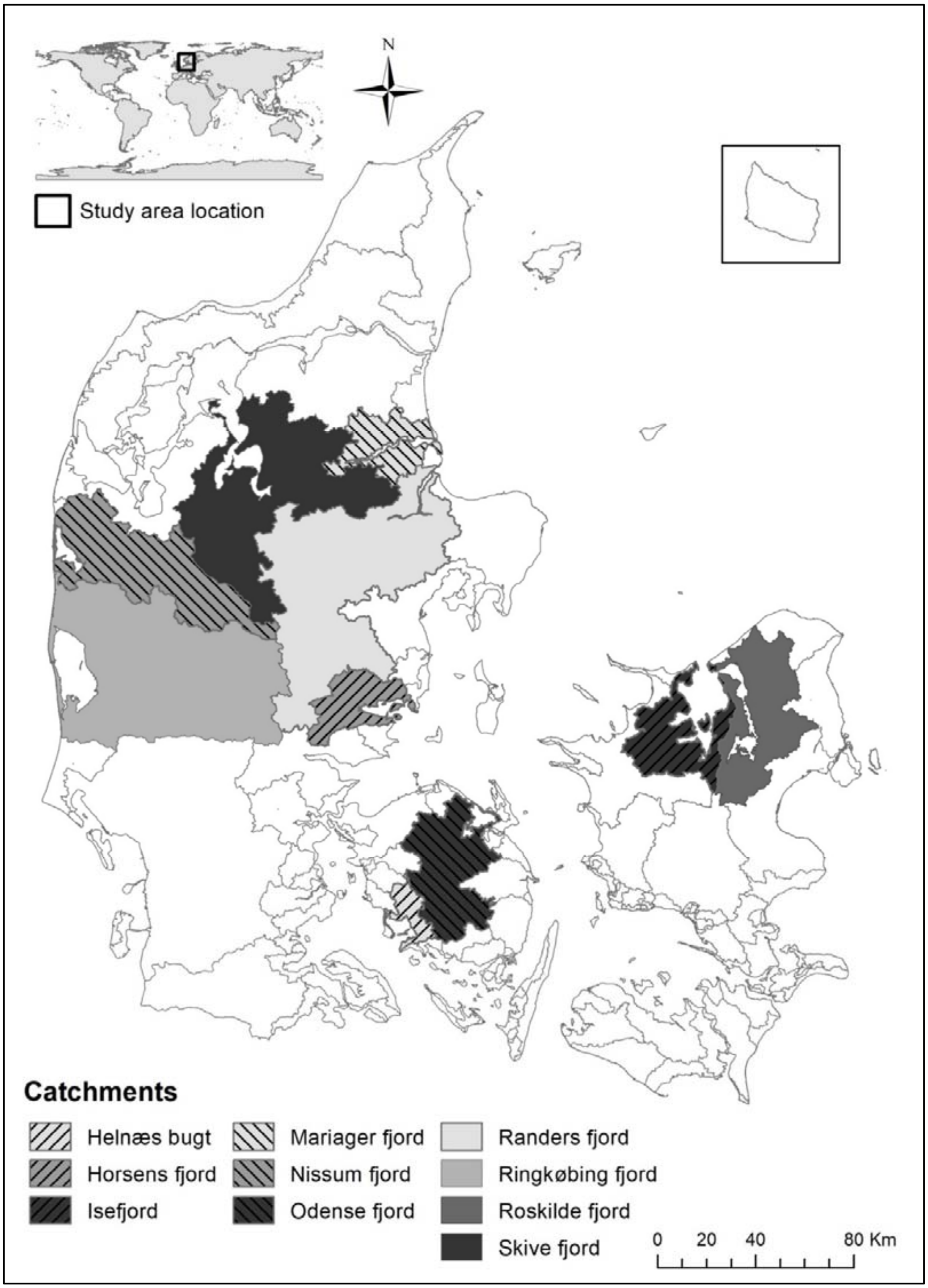
Abstract

We estimate economic benefits for seaside recreation and waterfront property when reducing nitrogen leaching to coastal water bodies. We apply impact pathway and benefit transfer methodology, linking total nitrogen concentration to water clarity (Secchi-depth). Ten catchments are analyzed comparing results for 2010 to a policy scenario that complies with the EU Water Framework Directive. The scenario reduces leaching with 5,200 ton N, downstream discharges to estuaries by 35% and provide significant Secchi-depth improvements. Our integrated assessment predicts an annual economic benefit for local residents of €35 million, and co-benefits of up to €57 million. Benefits are catchment-specific and differ for downstream discharges from €1 to €32 per kg N, while for upstream discharge losses they range up to €10 per kg N. When expressed per unit of farmland the policy scenario displays economic benefits spanning €8-176/ha. The span reflects the different physical, biological and human circumstances of each catchment.

Keywords

Impact pathway analysis; Benefit transfer; Water Framework Directive (WFD); Nitrate leaching; Secchi-depth



Map 1: River basin catchments designated in Denmark under the EU's Water Framework Directive, with indications of the ten study catchments.

Introduction

Article 4 of the European Union's (EU) Water Framework Directive (2000/60/EC; WFD) prescribes the environmental quality objectives for water bodies in EU Member States, i.e. 'good ecological status' for ground- and surface waters and 'good ecological potential' for heavily modified water bodies. Aiming for less stringent objectives is permissible only where "*the achievement of these objectives would be infeasible or disproportionately expensive*". Attainment of WFD objectives can be delayed where "*completing the improvements within the timescale would be disproportionately expensive*". River basin management plans (RBMP) must provide appropriate justifications for any such delay.

An EU guidance document explains the key role the proportionality principle should play in the economic analysis required by the Directive. While an appended information note maintains that estimates of monetary benefits are a must for water management, it cautions that the "*margin by which costs exceed benefits should be appreciable and have a high level of confidence*" and "*disproportionate costs should not begin at the point where measured costs simply exceed quantifiable benefits*" (ibid., p. 193). However, the EU document provides no further guidance on how to *"judge whether costs are disproportionate or not"* (WATECO, 2003:24).

In contrast to air pollution (Rabl et al., 2014), there is presently no conceptually coherent framework available to provide water managers with quantified estimates of the monetary benefits related to reductions in specific water pollutants[1]. In view of the WFD requirements this is an unfortunate lacuna, which our research seeks to address.

A main strand of monetary valuation research has addressed water quality improvements on the basis of contingent valuation (CV) surveys, resulting in estimates that are local and specific to the individual water bodies under examination, but which do not identify the marginal benefits of addressing specific water pollutants (Brouwer, 2008; Söderqvist and Hasselström, 2008, p.28). It is time consuming to conduct such studies for each water body, so improved techniques for the transfer of results across sites or catchments would be helpful in supporting such analyses (Eade and Moran, 1996; Navrud and Ready, 2007).

We present here a sequence of analytical steps, linking emissions to ecological water quality via changes in water clarity, as measured by Secchi depth.[2] Reference to water clarity as a proxy for ecological status is common among water managers. Despite its simplicity, Secchi depth provides a robust indicator, as it correlates statistically with seagrass depth limits, as well as with algae biomass and other organic compounds, that are stimulated by nutrient enrichment. .

Our focus is on a key water pollutant, nitrogen, which leaches from land use practices, and on its impacts on the water quality of estuaries as transitional water bodies (See review in the European Nitrogen Assessment, cf. van Grinsven, 2013). Previous research has established that the concentrations of total

[1] Van Grinsven et al. (2010) studied the potential costs related to drinking water contamination with nitrate from fertilizer surpluses. Von Blottnitz et al. (2006) reported costs related to atmospheric pollution with nitrogen, ammonia and greenhouse gases from mineral fertilizers. Both studies focus on human health effects, and do not address ecological water quality. Pretty et al. (2003) addressed only freshwater eutrophication. Van Grinsven et al. (2013) provide a wider assessment of nitrogen, with eutrophication as one element.

[2] Secchi depth as a measure of water transparency refers to the circular white Secchi disk introduced by Pietro Angelo Secchi in 1865.

nitrogen can explain 90 per cent of the variation in Secchi depth for the estuaries in question, while correlations with phosphorus is markedly weaker (Krause et al., 2007a, p. 122).

A baseline of deprived status (see Table 1) implies that improving ecological water quality requires long-term efforts, the economic values of which are likely to fade when subject to conventions of discounting in cost-benefit analysis. Our hypothesis is that by restricting the analysis to some short-term human preferences for clear water it should be possible to substantiate tangible economic benefits of nitrogen reducing measures that limit eutrophication. We aimed to explore how to use routinely sampled water data as the basis for monetary indicators that value reductions in nitrogen load.

The analysis is limited to economic benefits for which suitable valuation studies are available, such as the preferences for clear surface waters of recreational beachgoers, property owners, and residents adjacent to water bodies. The methodological approach of combining results from original valuation studies with transfer of benefits for a series of comparable water bodies is in line with a suggestion first made by Söderqvist and Hasselström (2008, p. 27). We close the analysis by considering some possible co-benefits, but make no claim to be comprehensive in covering all of the long-term environmental benefits from nitrogen reductions or from cleaner surface waters.

Materials and methods

The case material consists of ten catchment areas featuring large but shallow estuaries, which are customary in Denmark (see Map1; Conley et al., 2000). Over the past 30 years Denmark has implemented several nutrient action plans to protect water bodies from pollution (see Windolf et al., 2012a). Nevertheless, intensive farming with high fertilizer application rates (see Table 1) continues to cause leaching of nutrients, occasionally triggering eutrophication in estuaries and coastal waters, which suffer from inadequate water quality. According to estimates from the regional environmental centers, coastal waters receive about 30 per cent more nitrogen than WFD compliance allows for (Bredsdorff, 2015).

Estuary catchment	Region	Ecological water quality in year 2011[3]	Catchment area (km^2)	Utilised agricultural area (UAA) in 2011 (%)	of which sandy soils (%)	Surplus fertilizer (kgN/ha UAA)	Total discharge to estuary (Tons N)	Land use related discharge to estuary (Tons N)
Roskilde fjord	Zealand	Moderate/poor	1182	49	39	42	676	397
Isefjord	Zealand	Moderate/poor	770	65	44	42	948	846
Helnæs bugt	Funen	Moderate	184	65	74	51	206	190
Odense fjord	Funen	Poor	1060	60	53	48	1482	1254
Horsens fjord	Jutland E	Bad	520	65	38	53	865	720
Randers fjord	Jutland E	Bad	3255	59	71	54	2829	2376
Mariager fjord	Jutland E	Bad	572	63	95	59	900	818
Ringkøbing fjord	Jutland W	Poor	3477	61	91	82	4154	3646
Nissum fjord	Jutland W	Poor	1615	61	89	79	1951	1704
Skive fjord	Jutland N	Poor	2621	66	89	66	3363	3070

Table 1. Surface water quality, land use and farming practices in the 10 catchments analyzed, covering 35.5% of Denmark's land area, 36.5% of the utilized agricultural area (UAA) and 37.7% of farm nitrogen run-off (Sources: RBMPs, 2011; Windolf et al., 2012a; GEUS, 2015)

[3] 'Acceptable ecological status' is in a WFD context equivalent to High or Good status, while the unacceptable 'impaired ecological status' is equivalent to Moderate, Poor or Bad status.

We conduct an impact pathway analysis, which covers the sequence of events connecting burdens to impacts (Rabl et al., 2014). It features four steps, each requiring substantial amounts of data and analysis for appropriate representation. The methodology enables spatially differentiated results, whereby damage costs vary by site and receptor.

In relation to water bodies the four steps are as follows; 1) accounting for surplus nitrogen losses to the rootzone and resulting emissions to water bodies; 2) accounting for the transport, dispersion and resulting concentration changes in water bodies; 3) identifying adequate dose-response relations between nitrogen concentrations and water clarity and related impacts; and 4) applying monetary values to impacts (Figure 1). According to Hanley and Spash (1993, p. 203) the priority impacts from nitrogen are eutrophication and drinking water contamination. As monetary indicators for the health impacts of nitrogen in drinking water have been demonstrated elsewhere (van Grinsven et al., 2010; Andersen et al., 2011) we here focus on surface water eutrophication impacts. However, we do illustrate some possible co-benefits in Table 6.

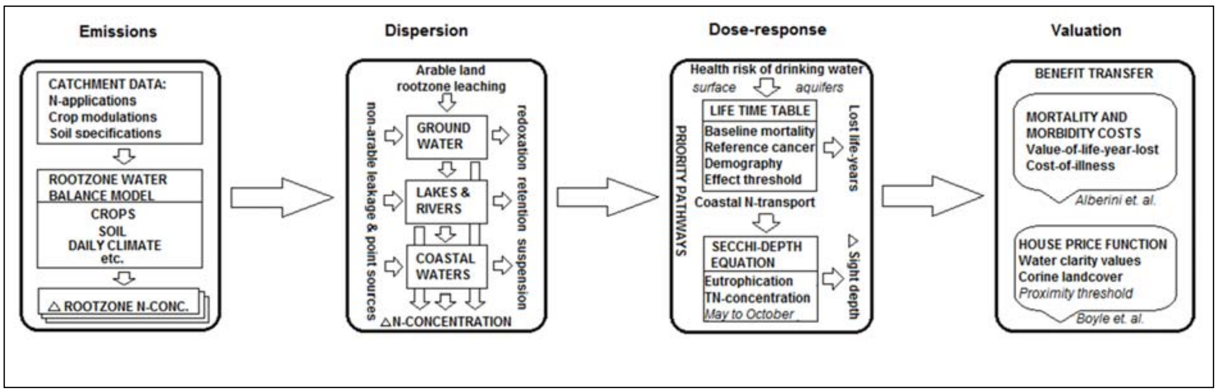


Figure 1. Impact pathway sequence for marginal costs of nitrogen leaching (Andersen et al., 2011).

The Danish national monitoring program NOVANA has accounted for nitrogen emissions to surface waters on an annual basis for more than twenty years, with specifications of direct discharges, land use leaching (including farming), atmospheric depositions and other sources (Windolf et al., 2012a). Nutrient concentrations of water streams are monitored and, with gauged water flows, allows for the calculation and estimation of waterborne discharge loads to estuaries. There is regular monitoring of surface water quality, with sampling of total nitrogen, suspended solids, chlorophyll, and measures of Secchi depth etc.

The point of departure for step 1 and 2 of the present analysis is Windolf et al. (2012a) who reported annual water stream nitrogen loads as well as nitrogen concentrations for the ten estuaries for a 20-year period (1990-2009). On the basis of these data, Windolf et al. (2012a, Table 4) derived useful statistical dose-response relationships for the average annual nitrogen concentrations in estuaries as a function of the respective catchment total nitrogen loads. This is expressed as the discharge-weighted total nitrogen concentration in inlet freshwater (see Table 2). These dose-response relationships allow for an assessment of the implications for annual total nitrogen concentrations in the upper and middle reaches of the ten estuaries as a function of changes in nitrogen loads. It is possible to disentangle leaching from the point sources of nitrogen. Detailed data for the individual estuaries are available in Windolf et al (2012b).

Providing a point of departure for step 3 in the present analysis, Carstensen (2009) established dose-response relationships for average Secchi depth as a function of the respective average annual nitrogen concentrations in a typical Danish estuary (Figure 2). Since the different fractions of light attenuating components in the water are not measured regularly, Carstensen (2009) used estuary total nitrogen (TN) as a proxy for all particulate and dissolved substances. He also identified, by non-linear regression analysis, the statistical relationship between Secchi depth (annual and seasonal) and TN, based on Lambert-Beer's law of light attenuation[4]. The data comes from 30 years of measurements at eight monitoring stations along an estuary nutrient gradient. The derived dose-response relation has an uncertainty factor of 0.5 m for 90% of the variation.

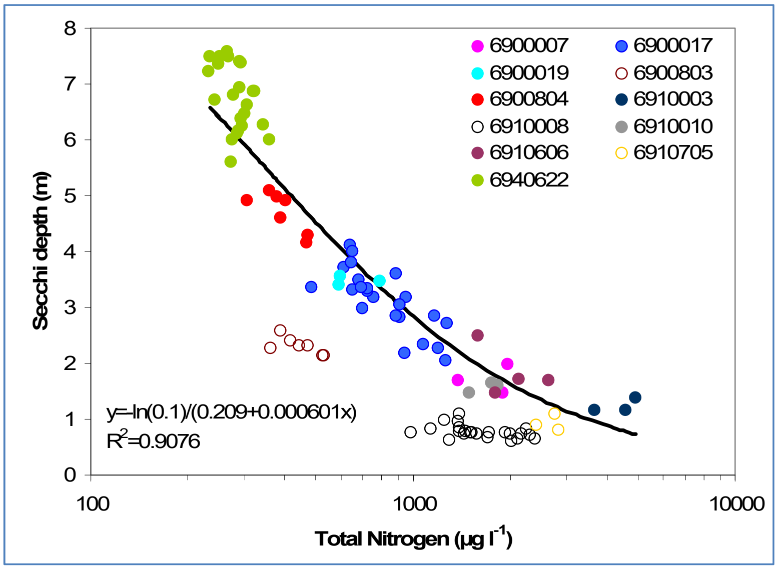


Figure 2. Relationship between annual TN and summer Secchi depth for 8 monitoring stations along a nutrient gradient in Odense Fjord 1976-2008. The standard error of the model was ±0.52 m. (Carstensen, 2009).

The Secchi-depth changes inferred for individual estuaries enable predictions of the willingness to pay (WTP) for emission reductions among waterfront property owners, beachgoers and recreationists. Several studies have demonstrated how property prices are influenced by water clarity, when properties are located adjacent to water bodies (Ara, 2007; Bin and Czajkowksi, 2013; Boyle et al., 1999; Gibbs et al., 2002; Tolun et al., 2012; Walsh et al., 2011). While such studies have a long tradition in North America, recent studies confirm similar impacts in Nordic countries (Vesterinen et al. 2010; Artell and Huhtala, 2016). Another strand of studies have demonstrated how the recreational value of surface waters is contingent on water clarity and influences the willingness to pay for cleaner surface waters (Söderqvist and Scharin, 2000; Atkins and Burdon, 2006; Atkins et al., 2007; Vesterinen et al., 2010). We used these studies for step 4 of our analysis, which requires valuation estimates for cardinal scale improvements in water clarity for waterfront property owners, beachgoers, and residents.

The main basis for the property price function is the study by Gibbs et al. (2002) conducted in the state of New Hampshire in USA. Although the water bodies at the study site are of freshwater, they are comparable in size to the estuaries examined here, which, as transitional water bodies, feature blends of freshwater

[4] Relationship between annual TN and annual Secchi depth: y=ln(0.1)/0.167+0.000712*TN (Carstensen, 2009).

and seawater. A more recent US study with a comparable focus on the relationship between property prices and water clarity addressed estuaries and lagoons. For these transitional water bodies it confirms that Secchi depth had a significant positive association with property prices in all model specifications, and thus was the strongest influencer of all water quality variables (Bin and Czajkowski, 2013). The mean WTP of 3.8% reported (ibid., p. 56) is slightly higher than the WTP transferred from Gibbs et al. (2002), which implies that property prices are assumed to increase with 3.5% per meter of Secchi depth improvement at our sites. This estimate was obtained with OECD's eight step procedure for benefit transfer (OECD, 2012), while applying the income elasticity identified by Barbier et al. (2016).

Artell and Huhtala (2016) explore the relationship between property prices and the ecological status for waterfront holiday homes in Finland, covering both seashore and lakefront properties. They find a decrease of 11.5% in prices where water quality status is below 'good' and a decrease of 16.7% when 'poor' (in Finland termed 'passable') for properties <250 m from the waterfront (ibid., Table 5). Their analysis demonstrates how owners' WTP, when related to technical WFD-measures such as Secchi-depth, provides a more conservative benchmark, in comparison to when WTP is related to owners' subjective assessment of water quality. We transfer the findings from Finland via the EQR (ecological quality ratios) calculated from predictions for the depth distribution of seagrasses for the various water quality classes (cf. WFD annex V). This gives us an estimate for the property price influence on holiday homes achieving a better quality class.[5] It implies a price premium for holiday homes if EQR improves sufficiently for the surface water to reach a new quality class; for example +1.2% premium for reaching 'moderate' status (EQR of 0.5); and +6.6% for reaching 'good' status (EQR of 0.74), on top of the 3.5% for all waterfront properties.[6] There will be a delay of several years between increased Secchi depth and improvements in the depth limits for seagrass vegetation, and thus ecological water quality. We therefore assumed a 10-year lag, and applied a discount rate of 2.5% in accordance with the Ramsey formula recommended by the European Commission (EC, 2014).

Few properties are located on the waterfront at our sites, due to the 1937 mandatory beach protection line of 100 meters which was extended to 300 meters in 1999 (Anker et al., 2004). We therefore considered properties within 300 meters of water bodies.[7] Information on permanent residence properties and holiday homes is derived from the Danish building and housing register (BBR) for the year 2012 (BBR, 2012). For each building unit, BBR contains information on location (xy-coordinates), use (e.g. residential, holiday homes) and on living space in square meters. We calculated the living space (which price statistics refer to) of permanent residence properties and holiday homes located in the waterfront zone for each estuary (cf.

[5] Considering that Finland has 380,000 waterfront holiday homes for a population of the same size as Denmark, the findings of Artell and Huhtala overall suggest that higher impacts of good water quality might be expected in Denmark (with 220,000 holiday homes) as homes at the waterfront are more precious.

[6] According to WFD ecological status is calculated as an 'ecological quality ratio' (EQR), being the ratio between observed value and reference value. Member States set the reference value for individual indicators. Classifications in coastal waters of Finland and Denmark both refer to seagrass depth limits.

[7] Walsh, Milon and Scrogin (2011) explore the proximity aspects of WTP for water clarity. While prices of waterfront properties increase with 4.1% per meter of Secchi depth improvement, they trace statistically significant impacts up to a distance of 1 km. The price impact ranges from 1.3-2.3% in the 100-400 m zone beyond the immediate waterfront. With properties spread equally across a waterfront zone of 300 m, the results of Walsh et al. suggest a property price impact of 2.8%, which is slightly less than the 3.3% of Gibbs et al. prior to benefit transfer. Ara (2007) also finds significant price impacts well beyond the waterfront itself, but in the absence of a good understanding of WTP distance decay, we use a 300 meters zone that extends only slightly beyond the immediate waterfront.

illustration in Figure 3). Properties in close proximity of the waterfront (<100 m) have been accounted for separately due to their upmarket value. Based on the welfare economic method of Møller et al. (2010) the 2016 property values have been amortized over 30 years and annualized. Municipal level data for prices per square meter of living space (2014-2016) come from the real estate Boliga database (Boliga 2017), while local adjustment factors for upmarket properties (i.e. within 100 m from waterfront) are based on real estate market data 2016-2017 provided by Boligsiden.dk (2017). The GIS tool BASEMAP accounts for the housing stock along the waterfronts of the respective estuaries and municipalities (Levin et al., 2012).

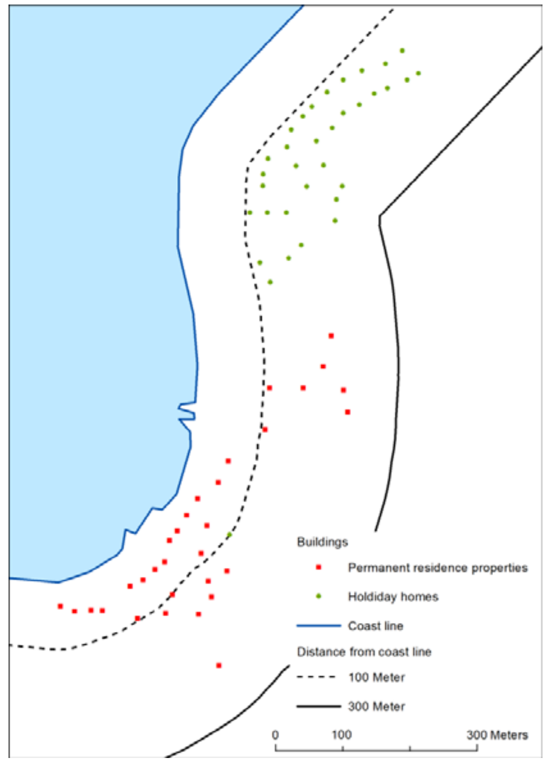


Figure 3. Waterfront property mapping, illustrative.

Apart from property-owners in the immediate vicinity of surface water bodies, estuaries provide use value to the general population with recreational opportunities. A recent survey shows that during summer months 41% of the population and 34% of foreign tourists head for the beaches for swimming (VisitDenmark, 2016). Beachgoers' WTP for improving water quality also increases per unit of improvement in Secchi depth (Söderqvist and Scharin, 2000). A travel cost study from Finland reports that reducing average water clarity by a meter leads to a loss of beachgoer benefits in the order of €29-87 million annually (Vesterinen et al., 2010). Most will visit beaches at the open sea, where water quality tends to be higher, but some do go to local beaches situated in estuaries. Nationwide monitoring of recreational site visits shows that 11% of beachgoers (4.5% of population) make use of estuary beaches (Jensen, 2003, p. 103ff). Data based on arrivals by car ensures there is no double counting of the WTP of waterfront property owners. We consider the resident population of the municipalities bordering the water bodies along with an estimate of the summer residents, based on the number of holiday homes and other tourist accommodation (camping sites, holiday centers, hostels and marinas) in the same municipalities (Statistics Denmark, 2017). Following the above procedure for adjusting study results to a policy site valuation estimate (OECD, 2012; Barbier, 2016) we conducted benefit transfer from an original study in Sweden (Söderqvist and Scharin, 2000) with the result that beachgoers' WTP is assumed to amount to €79 per meter of Secchi depth improvement. We assume that children are included under adults' WTP.

Atkins et al. (2007) conducted an original valuation study in relation to one of the water bodies here, Randers fjord, which is useful as a basis for inferring the WTP of remaining residents for amenity values of water clarity. They report that 22% of the local population make visits to the estuary to enjoy the views and the sound of the water. They surveyed respondents about how much they would be willing to pay for a program that would bring the estuary back into its pristine state over a ten-year period. Among all respondents, 69% were willing to pay (in 2003 prices) an average of €92 per year for an action plan to reduce nutrient inputs. The questionnaire mentions that it will involve "an increased transparency of 2.5-3 meters" throughout the estuary "so that the bottom will be visible while boating, swimming and fishing in most locations", allowing us to infer the WTP per meter Secchi. When adjusting for the higher WTP's of beachgoers and property owners, we find an annual amenity-WTP for all other adult residents (in 2016 prices) of €19.6 per meter Secchi depth, which we transfer across the sites.

Scenarios

Economic analysis implies that one must account for the marginal impacts of changes in emissions and water clarity from changes in environmental burdens. For this purpose two main scenarios are defined. A *reference scenario* holds 2010 data for annualized and water quantity weighted nitrogen transports to estuaries and the resulting concentrations of total nitrogen (TN) in the upper and middle reaches as a function of inlet concentrations. Reflecting the sensitivity of estuaries and corresponding to the stipulated reduction requirement for WFD compliance, a *policy scenario* involves a 35% reduction in farmland nitrogen leaching to the inlets.

Property prices and recreational benefits increase with improved water quality. The catchment-specific differences in water clarity, measured as Secchi depth, between the reference scenario and the policy scenario are used to determine the estimates of economic benefits from nitrogen pollution abatement. Our analysis assumes that WTP for waterfront properties increases by a fixed share of the base value per meter of improvement in Secchi depth. WTP of holiday-home-owners will increase slightly more when attainment of ecological quality ratios (EQR) for moderate or good surface water quality is within reach.

We use Windolf et al.'s (2012a) catchment-specific dose-response functions between estuary concentrations and inlet concentrations of total nitrogen to account for the reduced discharge burdens in the policy scenario. The resulting annual average estuary concentrations of nitrogen, referring to the upper and middle reaches, are in turn the basis for predicting the policy scenario induced water clarity changes on the basis of Carstensen's (2009) Secchi-depth dose-response relationship for estuaries. We use summer Secchi depth (May-October) as the measure most relevant for recreational benefits and annual Secchi depth as most relevant for property owners.

Benefit estimates related to units of nitrogen

We needed to obtain estimates of the marginal benefits related to the unit reductions in nitrogen loads. Assuming linearity, we estimate the downstream damage costs per unit of nitrogen discharged to estuaries by dividing the monetary estimates (that result from the differences between the reference scenario and the policy scenario) by the annual discharge load reduction of nitrogen. However, some mitigation measures address nutrient applications to crops and fields, making the upstream damage costs per unit of nitrogen leaching to the root-zone, as well as per unit of nitrogen applied to fields relevant too.

Upstream leaching to the root-zone with inorganic nitrogen is accounted for routinely with the empirical nitrogen leaching estimator model NLES, which uses field geography, crop data and a range of other parameters, weighted with water quantities according to a precipitation normal (Kristensen et al., 2008). Fertilizer data is available to NLES from the farmer plans compulsory for annual submission to the authorities. To obtain local scale leaching the NLES modelling provides estimates in a national matrix of 15 km^2 grid cells (GEUS, 2015).[8]

By aggregating the NLES local scale estimates to catchment level spatially in ArcGIS, water body specific figures for root-zone leaching becomes available. By dividing the policy scenario benefit per estuary with the tons of nitrogen lost upstream to the root-zone, catchment-specific benefits per unit of upstream nitrogen reduction at root-zone level are calculated. These results provide the basis for estimating the catchment-specific benefits per hectare of farmland. However, the NLES model may provide relatively conservative estimates of root-zone leaching, as it assumes that farmers comply with regulations and do not spread fertilizers in excess of norms on any parts of their lands. NLES accounts for losses in the short term (4-5 years), which is suitable for the approach here.

Results

Table 2 provides the mean long-run improvements in Secchi depth that arise, as well for the annual average as for the summer months average (May-October). With one exception, annual depth changes exceed summer depth changes. Despite a uniform policy scenario of a 35% reduction in nitrogen leaching, the implications for water clarity differ, due to differences in the area of catchments relative to the area of the estuary, as well as in features of the individual estuary. Reductions of fertilizer use over the large catchment land areas of Skive fjord and Ringkøbing fjord tend to provide greater improvements in water body clarity (3.4-3.9 m and 1.6-1.8 m respectively, Table 2), than the smaller land areas leaching to Horsens fjord and Isefjord (0.7-0.9 m and 0.4 m respectively, Table 2). Still, despite its small land area, significant water clarity improvement is predicted for Helnæs Bugt (1.1-1.4 m, Table 2), possibly due to short transport and residence times for nitrogen in its small catchment.

	Baseline scenario		Policy scenario 35% less N-leaching		Policy scenario implications			Secchi depth change induced by policy scenario	
Estuary catchment	N-inlet (mgTN/l)	N-estuary (ugTN/l)	Reduction in diffuse load (Tons N)	in % of total load	N-inlet (mgTN/l)	Estuary dose-response function (Windolf 2012)	N-estuary (ugTN/l)	Change predicted, annual (meter)	Change predicted, May-Oct (meter)
Roskilde fjord	3.27	814	138.9	21	2.60	$0,086N_i+0.533$	756	0.18	0.17
Isefjord	6.30	443	296.1	31	4.33	$0.027N_i+0.273$	390	0.41	0.35
Helnæs bugt	4.16	378	66.5	32	2.82	$0.095N_i-0.018$	250	1.39	1.13
Odense fjord	4.97	1062	439.0	30	3.50	$0.191N_i+0.113$	781	0.69	0.68
Horsens fjord	5.05	393	251.9	29	3.58	$0.062N_i+0.080$	301	0.88	0.73
Randers fjord	2.49	1083	831.8	29	1.76	$0.605N_i-0.423$	641	1.24	1.20
Mariager fjord	6.21	1099	286.2	32	4.23	$0.292N_i-0.714$	523	1.85	1.75
Ringkøbing fjord	2.64	752	1276.1	31	1.83	$0.713N_i-1.130$	174	1.82	1.62
Nissum fjord	2.70	1639	596.5	31	1.87	$0.562N_i+0.122$	1175	0.57	0.59
Skive fjord	5.28	1255	1074.6	32	3.59	$0.469N_i-1.221$	464	2.46	2.33

Table 2. Long-run Secchi depth changes expected with policy scenario.

[8] The root-zone loss of nitrogen refers to 2011 data from GEUS (2015), whereas the final estuary discharge from Windolf et al. (2012b) refers to 2010, but changes in fertilizer practices have been minimal between these two years.

Estuary catchment	Water-front housing stock value (<300m) (million €)	of which (a) <100m (b) holi-day homes (a) (%)	(b) (%)	Annualised housing stock value (<300m) (million €)	Annual WTP per meter Secchi depth change (million €)	Annual WTP for achieving EQR class 'good' (million €)	Annual WTP for achieving EQR class 'moderate' (million €)	Policy scenario: Annual WTP for 35% leaching reduction (million €)	Policy scenario: Annual WTP for EQR class upgrade due to 35% red. (million €)
Roskilde fjord	1,786	32	18	130	4.57	1.25	0.24	0.82	-
Isefjord	1,373	31	32	100	3.51	1.66	0.31	1.43	-
Helnæs bugt	182	36	62	13	0.47	0.38	0.07	0.65	0.07
Odense fjord	300	28	7	22	0.77	0.09	0.02	0.53	0.02
Horsens fjord	591	34	4	43	1.51	0.07	0.01	1.33	-
Randers fjord	84	5	5	6	0.22	0.02	0.005	0.27	-
Mariager fjord	277	33	5	20	0.71	0.05	0.01	1.31	0.05
Ringkøbing fjord	472	31	42	34	1.21	0.69	0.13	2.20	0.69
Nissum fjord	85	39	39	6	0.22	0.13	0.03	0.12	0.03
Skive fjord	351	22	49	25	0.90	0.67	0.13	3.49	0.67
								12.13	1.50

Table 3. Estimate of property-owners willingness to pay for improved water clarity.

Estuary catchment	Resident population (N)	Additional summer residents (N)	Estuary beachgoers, adults (N)	Beachgoer WTP per mSecchi depth change (summer) (million €)	Residents' amenity WTP per mSecchi for policy scenario leaching reduction (million €)	Beachgoers' annual WTP for 35% leaching reduction (million €)	Residents' annual amenity WTP for 35% leaching reduction (million €)
Roskilde fjord	133,851	99,015	9,015	0.86	1.12	0.15	0.19
Isefjord	153,256	165,806	12,555	1.25	1.30	0.43	0.45
Helnæs bugt	93,298	12,168	3,874	0.33	0.91	0.37	1.02
Odense fjord	190,103	1,007	6,884	0.55	1.86	0.37	1.26
Horsens fjord	116,992	37,727	5,810	0.52	1.08	0.38	0.79
Randers fjord	85,698	41,163	5,403	0.49	1.00	0.59	1.20
Mariager fjord	90,338	23,729	3,684	0.33	0.68	0.58	1.19
Ringkøbing fjord	108,604	132,321	9,519	0.95	0.99	1.54	1.60
Nissum fjord	78,851	33,705	4,267	0.39	0.78	0.23	0.46
Skive fjord	179,441	42,626	8,265	0.72	1.71	2.45	5.84
Sum	984,561					7.09	14.00

Table 4. Estimate of WTP of beach-goers and of residents' amenity values from improved water clarity.

Estuary catchment	Sum of WTP for property-owners, beachgoers and other residents (million €)	Leaching reduction to estuary (tons N)	Unit benefit per kg N reduced to inlet (€)	Fertilizer surplus reduction to rootzone (tons N)	Unit benefit per kg N reduced to rootzone (€)	Surplus fertilizer reduction (kgN/ha)	Unit benefit per hectare of farmland with policy scenario (€/ha UAA)
Roskilde fjord	1.16	139	€8.38	855	€1.36	15	€20
Isefjord	2.31	296	€7.81	735	€3.15	15	€46
Helnæs bugt	2.11	66	€31.68	214	€9.82	18	€176
Odense fjord	2.18	439	€4.96	1,063	€2.05	17	€34
Horsens fjord	2.49	252	€9.89	634	€3.93	19	€73
Randers fjord	2.05	832	€2.47	3,635	€0.56	19	€11
Mariager fjord	3.13	286	€10.93	753	€4.15	21	€86
Ringkøbing fjord	6.03	1,276	€4.73	6,128	€0.98	29	€28
Nissum fjord	0.81	597	€1.35	2,736	€0.30	28	€8
Skive fjord	12.45	1,075	€11.59	3,959	€3.15	23	€72
Sum	34.72	5,258		20,713			

Table 5. Unit benefits of nitrogen reductions in discharges, surplus and farmland balance.

Estuaries with greater water exchanges with coastal waters (Horsens fjord, Helnæs bugt) seem to be subject to marine inflows, reducing summer depths changes somewhat more compared to annual changes of Secchi depths. Catchment-specific dose-response relations between annual discharge and annual concentrations of nitrogen are complex and to disentangle these goes beyond our purposes here.

When combining the derived Secchi depth unit values with the results reported in Table 2, we obtain catchment specific estimates for the waterfront property owners' WTP for emission reductions (see Table 3). Due to preferences of holiday homeowners, Table 3 shows an additional WTP where Secchi depth changes would cause an estuary to shift to a higher ecological quality class, which depends on the EQR score. Marine vegetation with seagrasses is dependent on Secchi depth and is a core parameter for EQR. However not all the estuaries are likely to experience sufficient improvements from 2010 median depths under our uniform reduction scenario.

When combining the derived Secchi depth unit values with the results reported in Table 2, catchment specific estimates of recreational WTP for emission reductions is obtained (see Table 4). Not surprisingly, high beachgoer-WTP's result for estuaries that have many summer residents (Isefjord; Ringkøbing fjord), depending on the scale of expected improvement in Secchi depth. Table 4 shows also the WTP that results for amenity values of cleaner waters enjoyed by residents who are not waterfront property owners or considered beachgoers, with catchment differences reflecting mainly population densities.

Table 5 brings together the WTP of property-owners, beachgoers and residents at large, and presents the benefits than can be derived from the analysis. Due to the naturally occurring retention of surplus nitrogen during the transport from root-zone to surface waters, the spatial differentiation is essential to a proper understanding and use of unit estimates.

Total benefits for all 10 estuaries amount to €35 million annually for an aggregate reduction to surface waters of 5,200 tons N. High downstream unit benefits of about €10 per kg N_{inlet} for emission reductions result for several water bodies, with Helnæs bugt providing benefits as high as €32. Due to the logarithmic nature of the Secchi depth function (Figure 2), one unit of emission reduction in less polluted water bodies yields a relatively higher impact on water clarity. Low unit benefits of €1-2 per kg N_{inlet} are found for Nissum fjord and Randers fjord, both in Jutland, a result of their large catchment area, desolate location and poor water quality status. Medium unit benefits of €5-8 per kg N_{inlet} characterize the remaining estuaries.

When considering unit benefits per hectare of farmland they echo the low WTP score for Nissum and Randers fjord, whereas differences among the remaining catchments decline somewhat. On the predominantly loamy soils on the islands of Zeeland and Funen, the many arable farmers leach less surplus nitrogen compared to the sandy soils in Jutland where there is more manure surplus from livestock (see Table 1, column 7). These circumstances help offset differences in settlement and population densities as well as differences in baseline water quality, whereby several catchments from both regions offer unit benefits around €20-30 per hectare under the policy scenario. Four estuaries stand out as offering exceptionally high benefits of more than €70/ha; Mariager fjord, Skive fjord, Horsens fjord and Helnæs bugt. Their water bodies are extensive in size relative to the catchment land area, and so the management of each hectare of farmland becomes more essential to water quality. Conversely for Ringkøbing fjord despite predicted improvements in Secchi depth; as it drains from a huge catchment area the WTP per hectare (€28) is lower.

With results differing by an order of magnitude, it is evident that a policy scenario which assigns the same relative reduction of 35% to all estuaries will not be likely to be cost-effective, a finding in line with basic theoretical expectations of environmental economics. Water managers will have to tailor reductions to the specific physical, biological and human circumstances of each catchment if they wish to aim for some degree of economic efficiency, as the economics of abatement will be reflecting all these factors.

Co-benefits of nitrogen reductions

In cost-benefit analysis, monetary indicators for the value of nitrogen reductions for surface water quality complement other benefits arising from the pathways and impacts of nitrogen in the wider environment. These additional, or co-benefits, arise from ammonia; nitrate in drinking water; and greenhouse gases. Table 6 provides an illustrative overview of these co-benefits. These include estimates of health costs related to air pollution with ammonia (Andersen, 2018); estimates for the potential health benefits related to reduced drinking water nitrate (Andersen et al., 2011, updated with OECD's 2012 estimate for the value of a statistical life); and estimates for the social costs of N_2O greenhouse gas emissions, based on a carbon price of €24/t$CO_{2\text{-eq}}$.[9]

For each of the ten estuaries these estimates add to the benefits of the policy scenario on the conservative assumption that reductions will target mineral fertilizers and not nitrogen in livestock manure etc. The latter would entail even higher benefits due to the associated evaporation and leaching rates.

It follows from Table 6 that there is a higher overall baseline for the benefits of reducing nitrogen emissions for all ten water bodies when co-benefits are included, though with the differences between them relating largely to the value of Secchi-depth reductions. For some of the catchments with large land-areas and low

Estuary catchment	Upstream surplus fertilizer reduction to rootzone (kgN/ha)	Value of Secchi depth increase (Tb.5) (€/hectare)	Value of ammonia reduction (€1.07/kgN) (€/hectare)	Value of less nitrate in drinking water (€0.41/kgN) (€/hectare)	Value of GHG/N_2O reduction (€0.11/kgN) (€/hectare)	Unit benefit per hectare of farmland with policy scenario (€/ha UAA)	Upstream unit benefit per kgN to rootzone (€/kgN_{rootz})	Total benefits with policy scenario (miilion €)
Roskilde fjord	24	€20	€26	€10	€3	€58	2.95	€3.6
Isefjord	24	€46	€26	€10	€3	€84	4.74	€4.4
Helnæs bugt	29	€176	€31	€12	€3	€223	11.41	€2.7
Odense fjord	27	€34	€29	€11	€3	€78	3.64	€5.2
Horsens fjord	31	€73	€33	€13	€3	€122	5.52	€4.4
Randers fjord	31	€11	€33	€13	€3	€60	2.15	€12.0
Mariager fjord	34	€86	€36	€14	€4	€140	5.74	€5.3
Ringkøbing fjord	47	€28	€51	€19	€5	€104	2.57	€22.7
Nissum fjord	45	€8	€48	€19	€5	€80	1.88	€8.2
Skive fjord	38	€72	€40	€16	€4	€132	4.73	€23.6
Sum								€92.1

Table 6. Unit benefits of nitrogen reductions per hectare of farmland for eutrophication, ammonia evaporation, drinking water nitrate and greenhouse gases.

[9] With an emission factor of 0.01 kg N_2O per kg fertilizer N and a conversion factor from N_2O to N_2O-N of 28/44. With a standard GHG equivalence of 298 CO_2 to N_2O, a CO_2-unit price of €24 per ton CO_2 (reflecting expectations for prices of emissions trading allowances) translates into 0.11 eurocents/kgN of mineral fertilizer. Non-domestic GHG emissions related to production of mineral fertilizer excluded.

unit benefits of Secchi depth (Randers, Nissum) the co-benefits are close to exceed those of improving water quality with an order of magnitude. The benefits of Table 6 sum to a total of €92 million per year for all ten catchments, of which €57 million refer to co-benefits, providing indicators against costs of possible measures.

Comparison with cost estimates

The costs of nitrogen abatement are regularly subject to analysis, with figures reflecting price fluctuations for agricultural products and technological developments (see Table 7). Up to now, the politically acceptable benchmark for costs of measures has been about €7/kgN_{rootz} (Hasler et al., 2016). The creation of wetlands has, due to low costs, become a popular measure to reduce nutrient flows to surface waters, and is subsidized by the government. Support from the EU's Common Agricultural Policy is factored into the estimates, and net of EU support the welfare economic costs of wetlands are actually €7 per kgN_{rootz} (ibid.). In contrast, Sweden's nitrogen tax has been able to incentivize better use of fertilizers, curbing mineral fertilizer use with 6% or 10,000 tonnes annually at a cost of just €0.10/kgN. For higher reductions, unit costs remain low at €0.60/kgN and competitive with other measures in a Swedish context (Konjunkturinstitutet, 2014).

Policy Measure	Nitrogen reduction per hectare, annual	Certainty of reduction	Target group costs (€/kgN_{rootz})	Welfare economic costs (€/kgN_{rootz})
Autumn harvest crops	12-45	***	+0.67-2.55	+0.81-3.36
Autumn harvest crops, with crop rotations	12-45	***	+21.07-31.68	+28.05-41.74
Intermediate crops	9-13	**	+4.03-4.83	+5.23-6.44
N-intensive crops, beets	12-45	**	-20.94- -15.30	-27.65- -20.40
Winter wheat, early sowing	5-8	**	-10.74-+7.25	-14.23-+9.66
Energy crops, multiannual	34-51	***	-6.04-+14.36	-8.05-+19.06
Set aside, temporary	35-58	*	+3.76-25.50	+4.97-33.96
Permanent set aside	50	**	+9.26-11.14	+12.21-17.45
Buffer strips	37-74	*	+6.31-12.48	+8.32-16.51
Forest planting	50	**	+6.71-20.54	+8.86-27.25
Periodic restrictions on soil cultivation	10	**	+0.13	+0.13-0.27
Foddergrass conversion, autumn restriction	36	*	+1.88	+2.42
Manure spreading, autumn restriction	1850tN	**	+1.61	+2.01
Mini wetlands	5-20	**	+2.82-23.22	+3.62-31.14
Wetlands	120-190	***	+4.16-4.43	+5.50-5.91
Mussel breeding	600-900	**	+9.40-13.02	+12.48-17.32
Seaweed cultivation	16	**	+77.18-108.05	+102.28-143.36

Table 7. Unit costs of rootzone nitrogen reductions on farmland (Eriksen et al., 2014).

Eurostat data shows that the gross nitrogen balance in Denmark has been reduced with more than 50 per cent over the two decades from 1990-1994 to 2010-2014. One key measure to achieve reductions was the legal obligation to fertilize about 10 per cent below the economic optimum, a concession from farmer organizations to avoid a fertilizer tax. According to independent estimates, it caused an economic loss to farmers of annually about €65-130 million (Jacobsen et al., 2013). The average surplus of 80 kgN/ha nevertheless remains among the highest in the EU. Still, to spur economic growth the government in 2015 opted for a 'paradigm change' in nutrient management, allowing farmers to raise incomes by increasing fertilizer use by 10-20 per cent. To comply with WFD and cap the associated nitrogen leaching, the government has announced a new management approach of regionally targeted nutrient control measures.[10]

Discussion

Benefit transfer methodology

We acknowledge that use of benefit transfer is a pragmatic and somewhat coarse methodology that is useful mainly in the absence of original valuation studies. However, original studies are resource demanding, and where a study exists of the relationship between water clarity and WTP, questions do arise about how to transfer results from study site to other sites. The use of benefit transfer methodology is adequate when the main aim is to establish relatively simple monetary indicators for the value of nitrogen pollution reductions. For the transfers we applied a standard methodology, which takes account of differences in income and price levels between countries, but not of any national differences in preferences per se for environmental protection.

Benefit transfer from the stated preference study in Sweden poses less of a problem in terms of possible context and preference differences as compared to US studies. Still, to explore the relevance of the Swedish transfer estimate for beachgoers we calculated the travel costs reported by respondents of the nature recreation survey (Jensen, 2003). The difference in travel costs for beachgoers to seashore and estuary beaches respectively, can be interpreted as a preference for the higher water quality at seashore. With rates for mileage and spare time in accordance with the unit costs conventionally applied in transport analysis (Transportministeriet 2016), the WTP of seashore beachgoers constitutes about €50/mSecchi. It translates into a benefit at national level of about €93-101 million/mSecchi, which despite differences in study design is a figure in the same order of magnitude as in the travel cost study from Finland.

Predicting Secchi depth

Secchi depths are fluctuating over the years and seasons, where light is attenuated by particulate and dissolved matter within the water column as well as by water itself (Beer et al. 2014; Nielsen et al., 2002). In the absence of high-resolution models for our estuaries, the use of statistical relations can provide a reasonable estimate of how water clarity responds to changes in nutrient loads. By relying on monitoring data over several decades and along a nutrient gradient spanning the range of nitrogen concentrations of our estuaries, Carstensen's dose-response relationship for Secchi depth is adequate for providing useful predictions of the implications of changes in nitrogen loads. Our approach is more timid than if using an alternative Secchi-depth relationship, derived from several different estuaries but over a shorter period (Krause-Jensen et al., 2007a, p. 122).

Boyle, Poor and Taylor (1999) explored threshold effects and non-linearities in the demand for water clarity, and suggest that losses in Secchi depth might entail relatively greater property price impacts when the water body transparency is less than 3 meters. Tolun et al. (2012) found that property price sensitivity doubles when Secchi depth changes are in the lower transparency range. At the reference site for benefit

[10] The European Commission expressed its concerns in a letter to Denmark (EU PILOT 8540/16/ENVI).

transfer (Gibbs et al., 2002) all water bodies have visibility of more than 3 meters, while some of our sites fall below this threshold (Nissum, Ringkøbing and Randers). As these three water bodies feature relatively low benefits, it would seem worthwhile to explore further the non-linearities before drawing too firm conclusions on the findings.

Eight of our ten estuaries are polyhaline (Dahl et al., 2005, p. 18) and connected to the interior Danish open waters (Kattegat), with comparable conditions for water clarity. Two estuaries classified as euhaline and situated in western Jutland (Nissum fjord and Ringkøbing fjord) discharge to and exchange with the North Sea, with salinities close to 30‰. Exchanges with the open sea is in both cases controlled by a sluice, which water managers use for additional inlet of more saline seawater into the shallow and nutrient stressed water bodies. These management practices have facilitated improved water transparency. Studies show a salinity regime shift after 1996 for Ringkøbing fjord (cf. Håkanson and Bryhn, 2008; Petersen et al., 2008). Still, this hardly influences the dose-response function for estuary nitrogen concentrations (cf. Table 2), as a reanalysis of Windolf's data for Ringkøbing fjord for the period 1997-2010 reveals[11]. Applying the alternative Secchi-depth relation of Krause-Jensen (2007a, p. 122) suggests greater Secchi-depth improvements from our policy scenario for both euhaline estuaries, in the case of Nissum fjord almost twice as high. Still, even a doubling of the unit value would not challenge the finding that benefits of nitrogen reductions in the Nissum catchment are among the lowest of all the water bodies within the framework of impacts. For Ringkøbing the alternative Secchi-depth relation would change estimates of unit benefits by less than 10%. These considerations nevertheless underline that water body specific dose-response relations for Secchi depth can improve derived monetary indicators.

Ecological quality criteria (EQR)

The WFD establishes that biological reference conditions shall be defined for the ecological status of the various water bodies, which is linked to four biological indicators; algae, macrophages, fauna and fish. Water bodies should attain at least good ecological status using the normative definitions in the Directive annexes for biological quality elements (BOEs) relevant to the given water body. For transitional and coastal waters, seagrasses are sentinel indicators for eutrophication while other indicators are used as quality parameters. Seagrasses constitute one of the very few groups of flowering plants that live in the sea and provide important benthic habitats for other organisms such as fish, shellfish and other marine animals. They grow in shallow water bodies such as inlets, bays, estuaries and saltwater lagoons, and the depth-distribution of seagrasses is an important indicator for ecosystem health in the coastal zones falling under WFD in Denmark (MIM, 2011).

Human activity affects the abundance of seagrasses in several ways and they have been in decline over the past century, reflecting human stress on coastal water bodies (Orth et al., 2006). Attenuating substances in the water column determine the light available to seagrasses at the bottom. Their depth distribution is largely determined by water transparency, which is influenced by the amount of nutrients in the water. However, improved water clarity is not the only condition to enable seagrasses and the associated ecological quality of surface waters to recover. Recent studies have shown that it will require an active

[11] Dose-response in Table 2, column 7 for Ringkøbing fjord of $0.713N_i$-1.130 changes to 0.648 N_i-1.102.

restoration effort to obtain the ecological quality classes that improvements in Secchi depth may allow for (Kuusemäe et al., 2016).

As mentioned above we estimated the EQR resulting from Secchi depth improvements by applying a medium to short-term response relation (i.e. over 10 years) (cf. Krause-Jensen et al. 2007b). This implies that the extended seagrass distribution is predicted to be 70% of the depth expected from the improved light availability per se. In other words, Secchi-depth needs to improve by a further 50% to provide an EQR ratio sufficient to obtain the ecological quality class aimed for (cf. Pedersen et al., 2014). Improvements are further inhibited where sediment C/N ratios are low, reflecting excess organic materials (ibid.; Krause-Jensen et al., 2011). The low or missing score for several estuaries for holiday homeowners' WTP reported in the final column of Table 3 needs to be understood against this background.

Relevance of revealed preferences

The European Nitrogen Assessment (ENA) reported a benefit range of €5-20 per kgN for reducing nitrogen to surface waters, based on a Baltic Sea study, which according to one author may represent an overestimate (van Grinsven, 2016). It nevertheless compares reasonably well with the results of up to €32 per kgN for our water bodies.

The benefits of reduced eutrophication damages per se (improved Secchi depth) amount to €35 million per year for our 10 catchments, with a population of about one million citizens. In comparison the recent CV study for a clean Baltic Sea by 2050 estimates a Danish WTP per one million inhabitants of €22 million for a comprehensive clean-up program targeting also phosphorus and other pollutants under HELCOM targets (Ahtiainen et al., 2014). Apparently, when the benefits are regional and into the distant future, they are less valued than if local and available in the short term.

When including the co-benefits, the present assessment has coverage comparable to the classical eutrophication benefit study of Pretty et al. (2003), except implications for biodiversity[12]. In their UK study of freshwater eutrophication, annual damage costs range from £75-114 (€107-163) million, covering impacts of both nitrogen and phosphorus. These estimates refer mainly to restoration costs, whereas in the present study hedonic and contingent valuation studies provide the basis. Thus, for Denmark with less than 10% of the UK population, aggregate benefits of €92 million arise from reducing nitrogen applications on 1/3 of its farmland and affecting merely a subset of its surface waters. The relatively strong preferences for clean surface waters in Nordic countries is nevertheless supported by the results from Finland cited above e.g. €29-87 million in annual benefits to beach-goers, for a meter improvement in Secchi depth.

Leaving the methodological valuation issues aside, the heterogeneity of water quality benefits that we have demonstrated here deserves further reflection. The analysis of ten estuaries with catchments that differ with regard to their extent, soil types and agricultural practices, shows how the benefits of reducing nitrogen loads are highly site-specific. They differ from €10 to more than €100 per hectare of farmland. Benefits per unit of nitrogen reduced, upstream as well as downstream, display similar differences. Policy makers and water managers therefore face the choice of taking measures across a broad front to slowly

[12] Bird life reflects water clarity (Petersen et al., 2008). The economic value of trout angling is substantial, but fish stock response to Secchi depth needs to be derived (Hasler et al, 2016). Some freshwater invertebrate species are at risk with elevated nitrogen concentration.

improve water quality everywhere, or giving priority to those water bodies where the value for money is highest. Helnæs bugt, with its relatively clean waters, is already an anglers' paradise. Focusing nitrogen mitigation measures on, say, Horsens fjord, Mariager fjord, and Isefjord, which have relatively limited land areas and high benefits per unit of rootzone N-reductions, would be more cost effective. Such focused efforts could generate more value for the investment as well as perhaps more enthusiasm for eutrophication abatement among citizens, farmers and policymakers.

Conclusions

This article has demonstrated how to develop site-specific monetary indicators for the value of reducing nitrogen leaching to surface waters by combining routinely collected water quality data for nutrients and Secchi depth with a benefit transfer approach based on original valuation studies. Monetary indicators should be of interest to water managers and authorities that need to screen the economic benefits of water pollution control as part of a broader analysis of the proportionality of costs incurred for the fulfillment of the European Union's Water Framework Directive.

The aim was to link economic valuation to insights established with hard data from the natural sciences, to pioneer a credible approach of tangible monetary water quality benefits, based on the impact pathway and benefit transfer methodology.

To cover more fully the relevant environmental benefits would require an exploration of further impacts, such as implications of nitrogen pollution for stocks of fish and birds. However, with the longer timescale required for restoring ecosystems, such biodiversity related pathways might not add significantly to our economic estimates, when discounting benefits arising far into the future. The wider impact pathways of nitrogen fertilizers considered here include human health impacts related to ammonia and drinking water contaminations, and contributions to global warming from the greenhouse gas N_2O. Our estimates of monetary benefits for seaside recreation and property, including the co-benefits, suggests that the economic value of reducing nitrogen flows can be substantial, even without considering biodiversity related impacts.

The analysis found total annual benefits of €35 million from reducing nutrient leaching by 35% to ten local water bodies in catchments covering 37% of Denmark's land area to meet WFD objectives. With the associated co-benefits of measures that would also target ammonia, drinking water, and greenhouse gases (GHG) the annual economic value would be €92 million.

Considering the unit costs per kg N lost, our results are in the same order of magnitude as figures cited in the European Nitrogen Assessment, despite differences in methodological approaches, while they surpass findings of a recent contingent valuation study relating to the Baltic Sea. The benefits downstream of reducing nitrogen flows to water bodies range from €1-32 per kg N_{inlet}, while nitrogen retention implies lower unit benefits of reductions upstream, ranging from €0.3-10 per kg N surplus lost to the rootzone. Differences reflect not only the proportion of the water body to the adjoining catchment, but also features of population and recreation densities in the various catchments that influence the economic valuation. When the possible co-benefits are included, benefits of nitrogen mitigation range from €60-223 per hectare of farmland.

Bearing in mind the careful wording of the WFD guidance document on economic analysis, the Danish government benchmark of €7 per kg N_{rootz} for acceptable abatement costs matches reasonably well with the tangible benefits of about €3-9 per kg N_{rootz} that we identified in most of the ten catchments. However, for a few catchment areas with smaller benefits, the proportionality of measures that involve unit costs at the benchmark would be questionable. Natural and socio-economic circumstances vary, causing a given pollution reduction to provide different economic value for different water bodies. The present Danish policy focuses on the cost side of the equation and pays little attention to the economic benefits and their spatial distribution. The indicators identified here, as well as the experience from Sweden where nitrogen reductions has been achieved at low costs during the years when the fertilizer tax applied, should encourage stakeholders to consider a wider array of policy instruments and cost effective measures for reducing water pollution from nitrogen leaching.

Acknowledgement

Financial support to the ECOECO project from a strategic research grant of Ministry of Environment and Food to Aarhus University. We are grateful to Birgit Daetz of Boligsiden.dk and our colleague Jens Würgler Hansen for data support, as well as to our discussant Kelly Jones at the 6th World Conference of Environmental and Resource Economics. The anonymous review provided essential criticism and guidance.

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