REPORT

Future Nutrient Load Scenarios for the Baltic Sea Due to Climate and Lifestyle Changes

Hanna Eriksson Hägg, Steve W. Lyon,
Teresia Wällstedt, Carl-Magnus Mörtz,
Björn Claremar, Christoph Humborg

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Abstract Dynamic model simulations of the future climate and projections of future lifestyles within the Baltic Sea Drainage Basin (BSDB) were considered in this study to estimate potential trends in future nutrient loads to the Baltic Sea. Total nitrogen and total phosphorus loads were estimated using a simple proxy based only on human population (to account for nutrient sources) and stream discharges (to account for nutrient transport). This population-discharge proxy provided a good estimate for nutrient loads across the seven sub-basins of the BSDB considered. All climate scenarios considered here produced increased nutrient loads to the Baltic Sea over the next 100 years. There was variation between the climate scenarios such that sub-basin and regional differences were seen in future nutrient runoff depending on the climate model and scenario considered. Regardless, the results of this study indicate that changes in lifestyle brought about through shifts in consumption and population potentially overshadow the climate effects on future nutrient runoff for the entire BSDB. Regionally, however, lifestyle changes appear relatively more important in the southern regions of the BSDB while climatic changes appear more important in the northern regions with regards to future increases in nutrient loads. From a whole-ecosystem management perspective of the BSDB, this implies that implementation of improved and targeted management practices can still bring about improved conditions in the Baltic Sea in the face of a warmer and wetter future climate.

Keywords Baltic Sea Drainage Basin · Nutrient transport · Population growth · Climate change · Eutrophication · Baltic Nest Institute

INTRODUCTION

The combination of future changes in both climate and lifestyle has a great potential to alter future riverine nutrient loads to the sea (Hägg et al. 2010). It is therefore important to consider a full spectrum of possible future changes within our modeling scenarios. This is especially true in regions sensitive to eutrophication like the Baltic Sea. The Baltic Sea offers unique challenges from a management perspective in that it faces increasingly high nutrient inputs in southern sub-basins while large climate changes altering the flux of fresh water in northern sub-basins. This results in a Baltic Sea ecosystem that is severely stressed (Graham 2004; Conley et al. 2009) and perched on the precipice of alternative futures.

Globally, and regionally with respect the Baltic Sea drainage basin (BSDB), myriad models have been used to predict future riverine nutrient fluxes from the landscape under different climate and socioeconomic scenarios. Many of these approaches can be considered coupled, complex hydrological and biogeochemical models with detailed process-based representations of the release and movement of nutrients through the landscape and subsequent transport through the riverine system. While there are various strengths and weaknesses to such detailed modeling approaches, a common shortcoming is the requirement of extensive input datasets and information about the landscapes being studied. With a large set of parameters, comes the potential of large uncertainty in estimates (e.g., Beven 2001) that can mislead assessments of spatiotemporal water flow patterns, for instance with regard to vegetation–atmosphere interactions (Lyon et al. 2008), as well as waterborne nutrient loads and their

possible abatement (Destouni et al. 2006; Gren and Destouni 2012). Hence, the case can be made for considering less complicated, more empirically based approaches requiring few input parameters not only to provide a complementary perspective of current and future nutrient loads but also to create a useful tool for studying a full range of future scenarios that can help in assessing the validity of our more complicated modeling approaches (e.g., Van der Velde 2013).

Such empirical approaches, while often considered simple, may also find a basis in our theoretical understanding of nutrient and discharge dynamics from natural systems (Basu et al. 2010). As previously shown in Smith et al. (2003) and Smith et al. (2005), population density and river discharge can, on a larger scale and to a first order, be seen as robust proxies for the flux of total nitrogen (TN) and total phosphorus (TP) from hydrologic catchments. Thus we have chosen to use this simple regression approach to assess the magnitude and trends associated with potential changes in riverine nutrient fluxes from major Baltic Sea sub-basins under future climate and population change scenarios. The goal of this present study is therefore not to develop detailed process understanding per se; rather, we seek to explore a range of scenarios and the potential uncertainty associated with future predictions. Further, we seek to characterize the relative influence of climate change versus lifestyle changes (brought about through consumption and population shifts) on future nutrient loads to the Baltic Sea. This is useful from a management perspective as it can help in constraining future targets within the bounds of predictability of our models.

METHODOLOGY

Modeling Nutrient Loads and Data Considered

Annual riverine nutrient loads from sub-basins within the Baltic Sea drainage basin (BSDB) were modeled in this study using the regression relationships presented in Smith et al. (2005). Based on that work, annual riverine nutrient loads (L) were represented as a simple function of a region's annual discharge (Q) [m^3] and human population (X) [heads]:

$$L = k + a \log(Q) + b \log(X) \quad (1)$$

Together, these two independent variables provide proxies for both the source (via the population) and the transport (via the discharge) of nutrients from a landscape. In this current study we have calibrated the regression coefficients (k , a , and b in Eq. 1) for the relationships from Smith et al. (2005) to sub-basins of the BSDB to model annual loads of TN [tons] and TP [tons].

Discharge and nutrient load data were taken from the NEST decision support system (Wulff et al. 2007). These data have been derived from the Baltic Environmental Database (<http://nest.su.se/bed.htm>) based on sampling stations within the BSDB obtained from various environmental agencies. See Mörtz et al. (2007) for more details on these data. In this study, we consider the discharge and nutrient load data spanning the period from 1970 to 2006 in the calibration/validation of the parameters in Eq. 1 with 1970–2000 treated as the calibration period and 2000–2006 treated as the validation period. In addition to these datasets, total country population data were obtained from the United Nation (UN) Food and Agriculture Organization Statistic Database (FAOSTAT) (FAOSTAT 2011). For countries where long-term data on population size were available (i.e., Sweden, Denmark, Germany, Norway, Finland, Poland), the time period 1961–2006 was considered directly in this study (which extends beyond the calibration/validation period but is useful when considering projections of climatic change). For other countries (i.e., those founded in the 1990s) data from 1992/1993 to 2006 have been considered. From this, long-term datasets were estimated using a backwards calculation approximation based on the UN Medium growth scenario for the 1961–2006 time period (UN 2004). For the Czech Republic and Slovakia, long-term datasets were estimated by splitting the population of Czechoslovakia assuming the same population distribution as after the split of that nation (i.e., 65.6 % in Czech Republic and 34.4 % in Slovakia). The country-wise population data were redistributed among the Baltic Sea sub-basins using the History Database of the Global Environment (HYDE) population distribution for 2005 (Klein Goldewijk et al. 2011) assuming that no major migrations within the countries have occurred in the studied time period.

For calibration of Eq. 1, the BSDB was divided into seven sub-basins and the coefficients of Eq. 1 were calibrated accordingly for various groupings of these sub-basins independently for both TN (Table 1) and TP (Table 2). These groupings were determined by an initial analysis that highlighted different relationships between annual discharge and population against observed nutrient load (Fig. 1—shown for TN but similar found for TP) and on observed levels of nutrient load (Mörtz et al. 2007). This is similar in procedure to the methodology outlined in Hägg et al. (2010). For each grouping of sub-basins, Eq. 1 was calibrated on the period of commonly available data. As such, the coefficients reported in Tables 1 and 2 are the results of calibration using discharge, population, and nutrient load data for the period 1970–2000. These calibrated models of the seven sub-basins were validated using the available data for the period 2001–2006 with model fits evaluated using a simple R^2 statistic and the root mean squared error (RMSE).

Table 1 Coefficients from calibration (over the period 1970–2000) and uncertainty from validation (over the period 2001–2006 including root mean-squared error (RMSE)) for total nitrogen (TN) load estimates using Eq. 1 based on grouping of the main sub-basins within the Baltic Sea drainage basin (BSDB)

Sub-basin	k	a	b	R^2	RMSE (tons)
Bothnian Bay (BB)	6.59×10^{-5}	0.621	0.331	0.81	4 837
Bothnian Sea (BS)	6.59×10^{-5}	0.621	0.331	0.80	7 739
Gulf of Finland (GF)	6.59×10^{-5}	0.621	0.331	0.94	12 554
Baltic Proper (BP)	5.64×10^{-3}	0.732	0.231	0.93	92 770
Gulf of Riga (GR)	5.64×10^{-3}	0.732	0.231	0.55	14 147
Danish Straights (DS)	8.00×10^{-5}	0.732	0.231	0.98	6 526
Kattegat (KT)	5.47×10^{-5}	0.731	0.230	0.75	10 493

Table 2 Coefficients from calibration (over the period 1970–2000) and uncertainty from validation (over the period 2001–2006 including root mean squared error (RMSE)) for total phosphorus (TP) load estimates using Eq. 1 based on grouping of the main sub-basins within the Baltic Sea drainage basin (BSDB)

Sub-basin	k	a	b	R^2	RMSE (tons)
Bothnian Bay (BB)	1.25×10^{-9}	0.950	0.300	0.87	324
Bothnian Sea (BS)	1.25×10^{-9}	0.950	0.300	0.86	900
Gulf of Finland (GF)	1.25×10^{-9}	0.950	0.300	0.75	897
Baltic Proper (BP)	6.01×10^{-6}	0.412	0.646	0.93	5950
Gulf of Riga (GR)	6.01×10^{-6}	0.412	0.646	0.55	447
Danish Straights (DS)	8.39×10^{-6}	0.412	0.646	0.83	950
Kattegat (KT)	6.01×10^{-6}	0.412	0.646	0.80	313

Fig. 1 Observed total nitrogen (TN) compared to observed annual stream discharge and observed population for the Baltic Sea drainage basin (BSDB) sub-basins during the calibration period of Eq. 1 from 1970 to 2000

Table 3 Summary of the climate and populations scenarios considered in this study. See text for descriptions of each scenario

Scenario	Climate model	Emission scenario	Model ensemble	Population trajectory	Consumption adjustment	Factor addressed
1	ECHAM5	A1B	#1	Pop_S		Baseline scenario
2	ECHAM5	A1B	#2	Pop_S		Natural variability
3	ECHAM5	A1B	#3	Pop_S		Natural variability
4	HadCM	A1B		Pop_S		Climate model
5	CCSM	A1B		Pop_S		Climate model
6	ECHAM5	A2		Pop_S		Emissions (high)
7	ECHAM5	B1		Pop_S		Emissions (low)
8	ECHAM5	A1B	#1	Pop_NS	APC	Population growth
9	ECHAM5	A2		Pop_S	APC	Consumption
10	ECHAM5	B1		Pop_NS	% Reduction	Targeted reductions

Modeling Future Nutrient Load Scenarios

Once calibrated, the functional relationships between population, discharge, and nutrient loads form simple population-discharge proxies for estimating future TN and TP loads to the Baltic Sea. Such proxies can be used to investigate the future potential ranges and trends in nutrient loads and the potential uncertainty associated with future change scenarios across the seven major sub-basins of the BSDb. In this study, this was done by generating various future hydroclimatic and population scenarios (listed in Table 3 and described in detail in the following sections) that were then used to estimate loads based on the calibrated functional relationship previously outlined. These scenarios allow us to gauge the relative role of climate and lifestyle on future inputs of nutrients to the Baltic Sea and to explore future trends. Trends in the future nutrient load scenarios were analyzed using the Mann–Kendall trend test, which is a non-parametric test based on order of observations. The test was suggested by Mann (1945) and has been extensively used with environmental time series (Hipel and McLeod 2005).

Hydroclimatic Scenarios

To generate future hydroclimatic scenarios, several climate models and scenarios were used to force the CSIM hydrologic model (Mörth et al. 2007). As such, future discharge is modeled using the CSIM hydrologic model and not climate models [which can have significant issues with regards to water budget estimates (e.g., Van der Velde 2013)]. The CSIM hydrologic model is a framework offering an extension of the Generalized Watershed Loading Function model (GWLf), a lumped-parameter model, which describes the hydrology and corresponding fluxes of dissolved constituents from a watershed. Whereas GWLF was initially developed, tested, and described as a model for temperate zone watersheds in North America

(Haith and Shoemaker 1987), CSIM has been developed explicitly to represent the flux of water and nutrients from the BSDb. In the current work, we rely only on CSIM's hydrologic modeling components. As a first step to generate future hydroclimatic scenarios, the CSIM model was calibrated to present day observations. This calibration (and subsequent validation) is similar to that presented in Mörth et al. (2007) with a brief overview of the data considered presented here for completeness.

Data on present day temperature and precipitation were taken from the European Observation (E-OBS) database (Haylock et al. 2008) for calibration. E-OBS is a European land-only re-analysis based on interpolation between meteorological stations to a regular grid. Daily mean temperatures and precipitation with the resolution $0.44^\circ \times 0.44^\circ$ (ca. 50 km) were considered in this study to correspond to climate projections (see following sections). These gridded data were then used to calibrate (and validate) the CSIM model for the 117 catchments draining the BSDb (Mörth et al. 2007). Explicitly, the CSIM hydrologic model was calibrated based on E-OBS forcing data using observed discharge data for the time period 1996–2000.

Validation of the CSIM hydrologic model was carried out for the time period 1990–1994 over which the model could close the water balance to about 8.2 % of annual discharge on average over all catchments (Mörth et al. 2007). Further work (e.g., Meidani 2012) demonstrates that the CSIM model's monthly residual error (averaged over all catchments in the BSDb) never deviates more than ± 2 % of the total flow per catchment considering simulation over the period 1970–2000. There is a tendency for biased seasonal-scale errors in discharge estimates particularly in northern catchments. The monthly residual errors, however, are more-or-less stable with no strong trends on average lending confidence to future scenarios of discharge modeled using CSIM. In this current study, the daily discharge for each catchment estimated by the CSIM model were summed to an annual basis over each of the seven

major sub-basins considered in this study. This made the calibrated CSIM modeling results compatible with the nutrient load model considered in Eq. 1 such that CSIM could be used to estimate future hydrologic scenarios given future climatic scenarios. As such, we adopt the calibrated CSIM model as a starting point for generation of future hydroclimatic scenarios in this current study.

Future climate scenario (Table 3) data were downscaled from the ECHAM5 model (Jungclaus et al. 2006; Roeckner et al. 2006), the CCSM model (Vertenstein and Kauffman 2004) and the Hadley Centre couple model (HadCM3) (Gordon et al. 2000). Climate data for the period 1961–2100 were retrieved from the Rossby Centre at the Swedish Meteorological and Hydrological Institute (SMHI) where they were prepared in connection with the ENSEMBLES project (Hewitt and Griggs 2004). As part of this project, these coupled atmospheric and oceanographic climate models (AOGCM) were forced by different emission scenarios. Three different CO₂ emission scenarios were considered corresponding to relatively low (B1), middle (A1B), and high (A2) emission scenarios (Nakicenovic et al. 2000). For all three climate models, the middle emission scenario was considered while the relatively low and high scenarios were considered only in the ECHAM5 model (Table 2). Further, for the middle A1B emission scenario, all three available ensemble runs of the ECHAM5 model were considered to provide some insight to potential influence of natural variability. These ensemble runs were in developed assuming three different initial conditions in the ECHAM5 model setup and, thus, reflect the influence of natural variability within the modeling framework. All resulting climate models and scenarios were then dynamically downscaled with the regional climate model Rossby Centre Atmospheric regional climate model (RCA3) (Kjellström et al. 2005). The regionally downscaled models were bias corrected for the BSDB based on the E-OBS dataset for the time period 1961–1990. While this method, the so-called delta change approach, assumes a constant correction throughout the simulation period to adjust the annual average precipitation and temperature estimates, it maintains the relative seasonal variations estimated by each of the climate models.

Population and Consumption Scenarios

For the time period 2008–2100 we have considered two scenarios describing future population changes within the BSDB (Table 3). In the first population scenario (Pop_S) a steady state was assumed such that there was no population change from 2007 onward. In the second population scenario (Pop_NS) the population size was assumed to follow the non-steady state UN Medium Population Growth Scenario (UN 2004). This second scenario allows for a general decreasing

population size as the current trends in population dynamics show fertility rates below the 2.1 child per fertile woman needed to provide a steady state (Espenshade et al. 2003).

In addition, we also considered scenarios where human consumption changed in the future. As such, individuals in the future population consume more than their present-day counterparts and, therefore, count as more than one individual head in the population. In these adjusted per capita (APC) scenarios we have assumed a linearly increasing consumption of animal proteins reaching 75 g per capita per day in the year 2100 following the approach in Hägg et al. (2010). This end target is equivalent to a medium protein consumption diet of about the same order of magnitude present-day BSDB developed countries [e.g., Sweden 71 g/cap/day (2007), Denmark 72 g/cap/day (2007)]. Under such a trajectory of changing lifestyle, the largest increase in APC is expected in the transitional countries such as Poland, Russia, and the Baltic States. Yearly data on present-day animal protein consumption were obtained from FAOSTAT for the time period 1961–2007. For countries not having data for the whole time period we have made some assumptions on past animal protein consumption. The Czech Republic and Slovakia have both been assigned the present-day daily consumption equivalent to the statistical data for Czechoslovakia. The areas corresponding to the Belarus, Russia, Estonia, Latvia, Lithuania, and Ukraine have all been assigned a present-day consumption equivalent to that of the USSR. These projections are simplifications of lifestyle changes in that they account for population and consumption directly and do not consider the potential of, for example, coupled agricultural land pattern shifts associated with such lifestyle changes explicitly.

To connect these APC consumption scenarios to the population scenarios considered, we have assumed that the average present-day consumption of a person in the year 2000 is one person equivalent (PE). The year 2000 was chosen since Eq. 1 was created using data from around the year 2000 (Smith et al. 2005). This allows us to scale a person in a country with less animal protein consumption as less than one PE while a person in a country with higher consumption as more than one PE relative to the BSDB average. These present-day PE values per country were then adjusted to future consumption scenarios following the previously outlined linear growth patterns. The future projections of consumption were thus used to adjust the future population scenarios. By using this approach we can create additional scenarios with future population sizes adjusted for potential future protein consumption corresponding to the steady population scenario and the non-steady population scenario.

By combining the above population growth and consumption scenarios with the climate scenarios and

modeling results from the CSIM model, we can produce numerous future projections of nutrient loads from the BSDB using the calibrated population-discharge proxy (Eq. 1). In this current study, we consider three main simulations representative of this range of future scenarios (Table 3). The first assesses the ‘baseline’ nutrient load change by combining the middle ECHAM5 emission scenario with the non-steady population scenario APC (that allows for smaller future populations in the Baltic region). The second is a ‘worst’ case scenario from an environmental perspective with the high A2 emission scenario combined with steady population scenario APC. The third is a ‘best’ case scenario from an environmental perspective with a low B1 emission scenario and non-steady population scenario. Further, to account for potential future technology improvements (Voss et al. 2011), this third scenario includes reductions considered representative of an aggressive strategy to reduce the nutrient loads coming from the BSDB. For this current study, drawing from Voss et al. (2011), this entails annual reductions of 0.05 % in both TN and TP loads for Bothnian Bay and Bothnian Sea, annual reductions of 0.30 % in TN loads and 0.40 % for TP loads, respectively, for Baltic Proper and annual reductions of 0.10 % in both TN and TP loads for Gulf of Riga, Gulf of Finland, Danish Straights, and Kattegat per year until the year 2100. Conceptually, this ‘best’ scenario is similar to applying targeted management strategies to achieve greater reductions in the southern regions of the BSDB.

RESULTS

Population-Discharge Proxy Calibration and Validation

The population-discharge proxies for nutrient loads from Smith et al. (2005) (Eq. 1) were successfully calibrated on the available TN load (Table 1) and TP load (Table 2) data for the years 1970–2000. With regards to validation using the data from 2001 to 2006, the models performed well with R^2 values ranging from 0.55 for the Gulf of Riga to 0.98 for the Danish Straights with respect to TN loads and 0.55 for the Gulf of Riga to 0.93 for the Baltic Proper with respect to TP loads. The average R^2 values were 0.82 and 0.93 across all sub-basins for validation on TN loads and TP loads, respectively. These validation fits were aided by grouping BSDB sub-basins with similar nutrient-discharge relationships similar to what was done in Hägg et al. (2010). With respect to TN loads, the regression relationships give a slight over-estimate for all sub-basins with a relatively large overestimate for the Baltic Proper sub-basin (Fig. 2). Regardless, the general agreement between model and observed TN loads was rather good (Table 1)

and consistent with an ideal 1:1 slope (Fig. 2) between estimated and observed TN loads.

For TP loads, the performance of the population-discharge proxy was generally weaker than that for TN loads. Again, across all sub-basins, there was a slight over-prediction of TP loads (Fig. 2). For Baltic Proper and Gulf of Riga, there was also a general lack of agreement between the proxy estimated loads and the observed loads over the validation period (seen by the lack of correspondence to the 1:1 slope). Still, given the simplicity of the proxy presented in Eq. 1, the agreement seen between predicted and observed nutrient loads were adequate to allow the population-discharge proxies to serve as potential first-order representations.

Estimated Changes in Nutrient Loads

All scenarios considered estimated an increase in both TN loads (Table 4) and TP loads (Table 5) to the Baltic Sea over the period of simulation ending in 2100 except for the ‘best’ case scenario from an environmental perspective (Scenario 10) which estimated reductions in the TN and TP loads to the Baltic Sea (as expected). Considering the individual sub-basins, the Baltic Proper and Gulf of Riga sub-basins saw small reductions in TN and TP loads across some of the scenarios considered in spite of the estimated increase in total loads to the Baltic Sea. These reductions were brought about by variability in the future climate projections both through natural variability in the ECHAM5 model (e.g., Scenario 3) initial conditions and comparing across the three global climate models considered. The reductions due to variations in climate models were about 2–4 % within these two sub-basins highlighting the potential regional differences to be expected from future projections made using GCMs. In general, there were variations between the nutrient load estimates using the various climate model projections (Scenarios 1–5). On average, however, the models converge with respect to their trends in predicting an increase in total load to the Baltic Sea over the next 100 years.

The potential impact of various future emission scenarios (Scenario 6 and Scenario 7) were smaller across the entire BSDB compared to the natural variability based on initial conditions in the ECHAM5 model (Scenarios 1–3). The natural variability impact ranged from 5 to 12 % for TN load increase over the entire period of simulation while 10 % increases were seen for both the A2 and B1 emission scenarios (here, Scenario 6 and Scenario 7, respectively). For TP loads, the natural variability due to initial conditions ranged from 7 to 17 % increases while there was between 14 and 15 % increase under the future emission scenarios considered. This, again, highlights the potential regional variations that can be expected across a large

Fig. 2 Nutrient loads estimated using the population-discharge proxy compared to observed loads for the validation period 2001–2006 for all sub-basins

Table 4 Estimated change in total nitrogen (TN) from 1981–2000 period average to 2081–2100 period average for sub-basins of BSDB. Top half of the table shows absolute change in tons and lower half shows relative change as percentage of 1981–2000 period average. The sub-basins here are Bothnian Bay (BB), Bothnian Sea (BS), Gulf of Finland (GF), Baltic Proper (BP), Gulf of Riga (GR), Danish Straights (DS), and Kattegat (KT) while the bold number show totals and averages over the entire Baltic Sea Drainage Basin (BSDB)

Scenario	1	2	3	4	5	6	7	8	9	10
BB	7 311	7 525	4 403	8 772	2 593	6 826	4 749	8 651	12 919	1 110
BS	9 210	9 095	3 609	8 020	3 250	8 023	7 232	10 652	15 549	3 155
GF	22 874	20 203	8 923	9 975	5 080	22 096	18 388	19 887	28 180	−4 341
BP	10 117	8 782	4 838	−255	−11 597	2 332	13 023	5 941	78 648	−123 600
GR	2 759	2 269	−1 316	−39	−1 894	1 868	2 553	−5 974	15 254	−18 926
DS	2 387	2 545	1 902	1 717	1 540	1 711	3 079	4 509	7 320	−1 686
KT	5 245	5 240	2 375	3 388	2 792	4 569	4 083	7 784	17 850	−2 677
BSDB	59 903	55 658	24 735	31 578	1 765	47 425	53 107	51 449	175 719	−146 965
BB	16 %	16 %	9 %	19 %	5 %	15 %	10 %	19 %	26 %	2 %
BS	18 %	17 %	7 %	15 %	6 %	16 %	14 %	21 %	29 %	6 %
GF	24 %	20 %	9 %	10 %	5 %	23 %	19 %	20 %	30 %	−5 %
BP	3 %	2 %	1 %	0 %	−3 %	1 %	3 %	2 %	20 %	−33 %
GR	4 %	4 %	−2 %	0 %	−3 %	3 %	4 %	−9 %	23 %	−30 %
DS	7 %	7 %	5 %	5 %	4 %	5 %	9 %	13 %	20 %	−5 %
KT	10 %	10 %	4 %	6 %	5 %	9 %	8 %	15 %	31 %	−5 %
BSDB	12 %	11 %	5 %	8 %	3 %	10 %	10 %	11 %	26 %	−10 %

Table 5 Estimated change in total phosphorus (TP) from 1981–2000 period average to 2081–2100 period average for sub-basins of BSDB. Top half of the table shows absolute change in tons and lower half shows relative change as percentage of 1981–2000 period average. The sub-basins here are Bothnian Bay (BB), Bothnian Sea (BS), Gulf of Finland (GF), Baltic Proper (BP), Gulf of Riga (GR), Danish Straights (DS), and Kattegat (KT) while the bold number show totals and averages over the entire Baltic Sea Drainage Basin (BSDB)

Scenario	1	2	3	4	5	6	7	8	9	10
BB	561	578	318	683	176	519	349	627	823	161
BS	668	666	236	589	214	579	515	735	935	314
GF	1 825	1 643	714	796	414	1 760	1 463	1 703	2 059	298
BP	668	581	322	−25	−779	150	862	414	5 006	−7 865
GR	137	114	−54	6	−81	95	128	−246	704	−765
DS	95	101	75	69	60	67	123	175	284	−41
KT	234	234	103	150	123	203	181	342	795	−78
BSDB	4 188	3 916	1 714	2 267	127	3 374	3 620	3 751	10 606	−7 976
BB	23 %	24 %	13 %	28 %	7 %	21 %	14 %	26 %	33 %	7 %
BS	27 %	26 %	9 %	23 %	8 %	23 %	21 %	30 %	36 %	13 %
GF	39 %	32 %	14 %	16 %	8 %	38 %	31 %	35 %	44 %	6 %
BP	3 %	3 %	2 %	0 %	−4 %	1 %	4 %	2 %	24 %	−40 %
GR	6 %	5 %	−2 %	0 %	−3 %	4 %	6 %	−10 %	28 %	−33 %
DS	8 %	9 %	6 %	6 %	5 %	6 %	11 %	16 %	24 %	−4 %
KT	12 %	12 %	5 %	8 %	6 %	11 %	10 %	19 %	38 %	−4 %
BSDB	17 %	16 %	7 %	12 %	4 %	15 %	14 %	17 %	33 %	−8 %

region such as the BSDB. Such variations appear to have a larger impact than, for example, global climate alterations brought about via various emission scenarios.

Further, we can compare the potential impact of shifts in the consumption habits of the future BSDB populations (i.e., compare Scenario 6 and Scenario 9) to assess the

influence of lifestyle alterations on nutrient loads. Clearly, there is a larger impact when consumption (lifestyle) habits change in addition to population changes. For TN loads, this amounts to a 16 % increase in nutrient loads comparing consumption changes to just population changes alone. The difference is 18 % for TP loads to the entire Baltic Sea. As

such, characterization of consumption differences expected in future projection scenarios is quite influential on estimated future nutrient loads. In addition, the population scenario adopted (Scenario 8) has an influence on how nutrient loads change in future projections. Again, even though both steady and non-steady population scenarios lead to expected increases in TN and TP loads over the entire BSDB, regional differences appear. Specifically, the Gulf of Riga sub-basin experienced reductions in nutrient loads under Scenario 8 where populations are allowed to reduce following a non-steady model. These reductions were about 9 % for TN loads and 10 % for TP loads from the 1981–2000 period average loads to the 2081–2100 period average loads for the sub-basin, respectively.

Taken together, the results of these scenario analyses indicate that lifestyle changes will have a larger potential impact on nutrient loads to the entire Baltic Sea relative to climatic changes (Fig. 3). This can be seen to vary, however, across the sub-basins when considering the change estimated between the periods 1961–1980 and 2081–2100. Here, we compare the relative increases under the most ‘aggressive’ emission A2 scenario projection brought about through climatic changes only (i.e., Scenario 6) to those expected solely due to the steady-state population scenario with consumption adjustments (i.e., the difference between Scenario 9 and Scenario 6). Clearly, northern regions (those draining into Bothnian Bay, Bothnian Sea, and the Gulf of Finland) can be highlighted as regions where the potential impact of climatic changes on nutrient loads are higher than changes brought about due to lifestyle shifts (assessed via population and consumption changes). For the more southern regions, however, the increases in TN and TP loads expected over the next 100 years due to shifting lifestyle clearly outweigh those expected due to climatic changes alone. This leads to, on average over the entire BSDB, a situation where the increases in nutrient loads due to lifestyles changes are expected to be greater than those brought about through shifts in the climate alone.

Trend Analysis of Future Nutrient Loads

The majority of scenarios considered in this study had significant trends for future TN and TP loads (Table 6) when looking across the entire period of simulation. Considering the TN and TP load trends, the northern most sub-basins (e.g., Bothnian Bay and Bothnian Sea) exhibited significant ($p < 0.01$) increasing trends across all scenarios considered. Looking across all the other sub-basins, the environmental ‘best’ case scenario (Scenario 10) showed significant decreasing trends in TN loads for the Gulf of Finland, Baltic Proper and Gulf of Riga. In the south-western most sub-basins (e.g., Kattegat and Danish

Fig. 3 Percentage increase in nutrient loads due to either climate change or lifestyle change across all sub-basins and the entire Baltic Sea Drainage Basin (BSDB) between the period 1961–1980 and 2081–2100

Straights), the negative trends seen in Scenario 10 were not significant under the Mann–Kendall test considered in this study. With regards to the TP load trends under Scenario 10, significant decreasing trends were exhibited for the Gulf of Riga, Danish Straights, and Kattegat. Clearly, there is an impact of moving from northern, low population density to southern high population density regions on the projected trends generated under this scenario. Again, similar to the previous comparisons, there were differences in significance of the future estimated nutrient load trends driven by the selection of climate model (Table 6). This was most clear in the Gulf of Riga, Baltic Proper, and Danish Straights sub-basins.

Considering the ‘worst’ and ‘best’ case scenarios from an environmental perspective (e.g., Scenario 9 and Scenario 10, respectively), we can see that, based on the simple population-discharge proxy either with population

Table 6 Theil slopes from a Mann–Kendall analysis of estimated future nutrient trends from 1980 through 2100. The top half of the table shows slopes for TN loads and the lower half shows slopes for TP loads with bold indicating significance at $p < 0.01$ level. The sub-basins here are Bothnian Bay (BB), Bothnian Sea (BS), Gulf of Finland (GF), Baltic Proper (BP), Gulf of Riga (GR), Danish Straights (DS), and Kattegat (KT) while the bold number show totals and averages over the entire Baltic Sea Drainage Basin (BSDB)

Scenario	1	2	3	4	5	6	7	8	9	10
BB	72	50	92	43	76	57	94	138	45	25
BS	80	55	94	57	78	58	102	153	45	22
GF	179	105	130	114	180	157	184	253	25	–51
BP	109	123	181	233	44	107	167	94	869	–1182
GR	24	28	6	16	15	24	23	–43	175	–184
DS	4	16	27	35	14	6	25	21	34	–10
KT	63	75	35	84	58	55	36	88	114	–18
BB	5.3	3.5	6.9	2.9	5.7	4.1	6.3	8.6	3.5	2.4
BS	5.6	3.7	6.7	3.8	5.4	3.9	6.6	8.9	3.3	2.2
GF	13.8	7.6	9.4	8.5	13.5	11.6	14.2	16.9	5.4	1.3
BP	6.7	7.5	11.5	14.7	2.2	6.3	10.5	6.1	6.7	7.5
GR	1.2	1.3	0.3	0.7	0.7	1.1	1.1	–1.7	8.0	–7.5
DS	0.7	1.0	1.2	1.5	0.9	0.8	1.2	2.8	4.4	–0.8
KT	1.7	1.9	1.3	1.9	1.6	1.6	1.2	3.3	6.6	–0.6

adjusted to account for consumption increase or with loads reduced due to potential future advances (i.e., reductions on the order of those estimated by Voss et al. 2011), it is possible to obtain both large increases and decreases in future TN loads (Fig. 4) and TP loads (Fig. 5) across the sub-basins of the BSDB. These scenarios provide relevant upper and lower bounds to future projections such as those considered in this current study (for example, the ‘baseline’ scenario represented in Scenario 8). Further, these upper and lower bounds can be considered useful to help constrain future analysis as better data or process representations come available with regards to population and consumption patterns and future trends.

DISCUSSION AND CONCLUDING REMARKS

Population and consumption adjustment per capita in combination with dynamic modeling of future climates (Table 3) provide a solid base for estimation of future nutrient loads to the Baltic Sea. This was achieved in this study by using a simple population-discharge proxy for nutrient loads such that consumption can be related directly to nutrient loads via adjustment of future population projections. Clearly, the results of the study highlight that the majority of future projections point to increased TN and TP loads coming into the Baltic Sea in the next 100 years (Table 6). This is in line with previous climate simulation based work (e.g., Graham and Bergström 2001; Reckermann et al. 2011).

A potential strength of the approach considered in this current study is the simplicity with which population and

hydrology (Eq. 1) together can account for future variations in lifestyle. This makes it possible to isolate impacts of lifestyle and climate and highlights how lifestyle changes potentially play a larger role in the increased nutrient loads relative to climatic changes (Fig. 3). This approach is advantageous as it leverages widely available and consistent datasets (like that available from the UN Medium Population Growth Scenario) to create estimates of future lifestyle and climate impacts that can be tailored to provide upper and lower limits to potential future nutrient loading. This is a valuable range as it provides a robust tool by which future scenarios drawing from more complex approaches (e.g., LPJ Guess from Smith et al. (2001)) can be benchmarked. This potentially allows for us to better constrain and assess predictions made by combinations of more complicated population models, dynamic lifestyle projections (e.g., specifically accounting imported/exported food) and nutrient source allocation/transport models (e.g., specifically accounting for agricultural changes in nutrient production and utilization) in a whole-systems approach.

Further, by coupling the simple population-discharge proxy for load estimate to complete dynamic GCM simulations, we can explore the potential influence of regional differences in climate projections across the BSDB. In the northern sub-basins, for example, there is strong influence of the climatic projection on future loading of TN and TP to the Baltic Sea (Fig. 3). This is seen by the larger range and magnitudes in future projections in northern sub-basins (e.g., Bothnian Bay and Bothnian Sea) across various climate projections relative to those seen in the more southern regions (Tables 4, 5). As such, these cold regions are potentially more sensitive to future changes in climate.

Fig. 4 Standard box plots indicating change in total nitrogen (TN) load (tons) between the period 1961–1980 and 2081–2100 for all sub-basins for Scenario 8 (black), Scenario 9 (dark gray), and Scenario 10 (light gray)

This is consistent with local (Lyon et al. 2010) to regional-scale simulations (Teutschbein and Seibert 2010). In addition, GCMs tend to diverge in these regions in their ability to project similar changes in future climates. The Arctic and Sub-Arctic are long-standing trouble spot for future projections (e.g., Boé et al. 2009). As future models improve performance in these regions, one would expect future projections of nutrient loads to correspond and uncertainty to be reduced. These sensitivities to climatic shifts in the northern regions are countered in the southern regions of the BSDB by increased relative importance of lifestyle on future nutrient load projections (Fig. 3).

Independent of the climate projections, there appears to be some regional variations in the performance of the population-discharge proxy for nutrient loads. The population-discharge proxies appear to perform less reliably in

southern regions (mainly Baltic Proper and Gulf of Riga) than in northern regions (Fig. 2). Several potential explanations can be put forward. This limited ability in southern regions may be, for example, due to the large number of concentrated people and/or the waste water treatment facilities in these areas. As such, waste processing (or lack thereof) potentially influences the robustness of the simple population-discharge load proxy. In these BSDB systems, however, the contribution of point sources (like waste water treatment facilities) to total loads are rather small and often contribute less than 25 % of the annual average loads of TN and TP (Mörth et al. 2007). This limited ability in southern regions could also be indicative of a decoupling of agricultural intensification from regional consumption leading to, for example, a weakening of the ability of population to serve as a nutrient load proxy. So, clearly,

Fig. 5 Standard box plots indicating change in total phosphorous (TP) load (tons) between the period 1961–1980 and 2081–2100 for all sub-basins for Scenario 8 (black), Scenario 9 (dark gray), and Scenario 10 (light gray)

there are potential issues (as is expected with such a simple proxy in an empirical setting) when extrapolating beyond the calibration period and limitations to the assumed connections (e.g., Smith et al. 2005) between consumption and nutrient loads.

The model calibration procedure considered also has potential influence on performance. In this current study, for example, calibration was applied to data covering the time period 1970 through 2000. As such, temporal variations in the number waste water treatment facilities and their efficiency were not explicitly considered. This could have impact in rapidly developing regions such as the Baltic States, Russia, and Poland that leads to poor model representation of nutrient loads. Further, in-stream retention of nutrients may influence the loads experienced at the

river outlets such that the rather large and often human-controlled flows in the southern sub-basins respond differently to their northern BSDB counterparts. This would be consistent with the results from Arheimer et al. (2012) where in-stream retention changes were expected in relation to future climatic changes. In addition, more recent regional reductions seen in nutrient exports [e.g., total land-based N load to Danish coastal waters has been reduced by ca. 50 % since 1990 (Kronvang et al. 2008)] may not be adequately reflected by adopting a calibration window from 1970 to 2000 (these impacts, however, are quantified under the validation period in Tables 1, 2).

These limitations present a potential shortcoming of the approach considered here that could be addressed in the future projections to some extent by the inclusion of

population growth dynamics (Smith et al. 2001), agricultural changes, higher order relationships to characterize consumption-nutrient relations, or more process-based modeling approaches. We characterize nutrient losses as a function of human population levels following the approach outline in Smith et al. (2005). This approach provides a good first-order assessment, however, fails to account for more dynamic changes that can potentially occur (and influence) on more regional scales such as those relevant for (at least part of) the BSDB. For example, agricultural development has occurred across large parts of the region with impacts on the water cycle (e.g., Jaramillo et al. 2013). Such influences are captured here by the use of a hydrologic model to estimate stream discharge, but the secondary impact on nutrient loading is not directly accounted for in the approach considered. Further, while population and hydrologic changes can be addressed in future scenarios, future agricultural development and changing production patterns due to, for example, economic and climatic forces are not explicitly considered. Agricultural land area remains rather stable (and contracts in some regions) in recent years over much of Europe while productivity has still increased due to new agricultural methods and technological improvements (Rabbinge and van Diepen 2000). More detailed work considering both relative role of the pattern of activities in the landscape and the pattern of water discharge (e.g., Hong et al. 2012; Meidani 2012) is clearly necessary to complement this approaches considered here.

The estimations presented here, however, have a clear utility from a management perspective as they can help in constraining future targets within the bounds of predictability of our more advanced and dynamic models. There is promise that this simple proxy captures first-order controls on potential impacts of future climate change and changes in protein consumption (albeit given assumptions about agricultural production, nutrient balances, and human consumption patterns) across much of the BSDB (Tables 4, 5). This result is promising as data limitations and the lack of consistent databases make it difficult to develop a completely process-based model capable of incorporating future population dynamics and consumption scenario testing at the scale of the BSDB. Further, considering nutrient fluxes from catchments, the inconsistencies on national load and source-oriented approaches to estimating nutrient loads to the Baltic Sea may lead to serious misinterpretations and development of inadequate management strategies (Mörth et al. 2007). To allow for comparison and application across such large geographical and geopolitical regions like the BSDB, models and prediction frameworks need to draw upon consistent data (Hannerz and Destouni 2006). Consistency between data and modeling frameworks is, thus, a necessity. Once

hydrologic models and/or management tools are established based on spatially and temporally consistent data environments, they can help develop more feasible representations of future scenarios and can allow for evaluation of model performance over a range of conditions and regions. This allows for explicit testing of modeling assumptions and identification of times and places where further process information may need to be considered (i.e., key areas for improvement). The robust population-discharge proxy presented here provides a relevant benchmark for evaluation of such modeling development from which nutrient transport and landscape management can be addressed (Wulff et al. 2007).

Consider, for example, that the maximum allowable N and P inputs to Baltic Sea annually are on the order of about 21 000 tons P and 600 000 tons N according to the Baltic Sea Action Plan (HELCOM 2007). The required reductions from present loads to achieve these goals are about 15 000 tons P and 135 000 tons N, respectively. While this considers the entire Baltic Sea, reductions of the loads to the individual basins are equally important. This is because the basins within the Baltic Sea are interlinked and, thus, influence each other. In view of the scenarios investigated in this current study, it seems that only Scenario 10 could give reductions of the N and P loads on the scale required to achieve current goals set forth in Baltic Sea Action Plan. The required reductions, however, still may not explicitly be achieved considering P loads to all basins individually but seem more possible for N loads to almost all individual basins. In general, this analysis indicates that a substantial reduction of livestock (e.g., Wulff et al. 2007) appears to be needed to fulfill the BSAP. At the same time, increases in number and efficiency of waste water treatment facilities would also likely have an effect (mainly on P) consistent with measure outline in Voss et al. (2011). The feasibility with regards to social-economic policy and governance behind the Scenario 10 is difficult to evaluate at this time but remains the focus of upcoming work.

Clearly, it is important to consider uncertainty in our modeling (regardless of the approach adopted) and its potential impact across large scales. As there was considerable variation in the climate scenarios considered in this study, future runoff and nutrient loads vary largely depending on climate scenario. There is a need to better assess and account for spatial variability of this uncertainty across the BSDB to properly consider the role it will play in the projection of future loads across the various sub-basins. Notwithstanding such uncertainty, the results of this current study indicate that changes in lifestyle have the potential to overshadow climate effects on future nutrient loads to the Baltic Sea (Fig. 3). This relative difference is not, however, uniformly distributed across the entire

BSDB. It is mainly the southern regions where lifestyle changes are predicted to outpace climatic changes with regards to increasing nutrient loads to the Baltic Sea. It is in these regions where we have a strong potential for identifying ‘hotspots’ of management for reducing nutrient loads. This is consistent with the environmental ‘best’ scenario presented in this study where higher reduction in nutrient loads are implemented in the modeling through significantly higher and targeted load reductions in, for example, the Baltic Proper sub-basin (Tables 4, 5). This leads to a potentially improved condition over entire BSDB (Figs. 4, 5). Such approaches to target management have been successful in other regions where excess nutrients are a concern (e.g., Walter et al. 2000; Lyon et al. 2006). This holds promise for the future of the coupled BSDB system from a whole-ecosystem management perspective where implementation of improved and targeted practices (e.g., Voss et al. 2011) can still potentially bring about improved conditions in the Baltic Sea in the face of a warmer and wetter future climate.

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AUTHOR BIOGRAPHIES

Hanna Eriksson Hägg is a postdoctoral researcher with the Baltic Nest Institute. Her research interests include large scale nutrient losses from catchments to recipient water bodies and nutrient source apportionment.

Address: Baltic Nest Institute, Baltic Sea Centre, Stockholm University, Stockholm, Sweden.

Steve W. Lyon (✉) is an associate professor at the Department of Physical Geography and Quaternary Geology, Stockholm University and a researcher with the Baltic Nest Institute. His research centers on how to best observe and represent hydrologic processes at multiple scales and the associated transport of nutrients.

Address: Baltic Nest Institute, Baltic Sea Centre, Stockholm University, Stockholm, Sweden.

Address: Department of Physical Geography and Quaternary Geology, Stockholm University, Stockholm, Sweden.

e-mail: steve.lyon@natgeo.su.se

Teresia Wällstedt is a postdoctoral researcher with the Baltic Nest Institute. Her research interests include biogeochemical cycling in the Baltic region.

Address: Department of Geological Sciences, Stockholm University, Stockholm, Sweden.

Carl-Magnus Mört is an associate professor at the Department of Geological Sciences, Stockholm University and a researcher with the Baltic Nest Institute. His research focus is on biogeochemical processes in terrestrial systems, especially large-scale transport and weathering.

Address: Baltic Nest Institute, Baltic Sea Centre, Stockholm University, Stockholm, Sweden.

Address: Department of Geological Sciences, Stockholm University, Stockholm, Sweden.

Björn Claremar is an associate professor at the Department of Earth Sciences, Uppsala University. He is currently working with assessing the distribution and change of depositions of acidifying and eutrophying compounds over the Baltic Sea catchment area.

Address: Department of Earth Sciences, Uppsala University, Uppsala, Sweden.

Christoph Humborg is an associate professor at the Department of Applied Environmental Science, Stockholm University and a researcher with the Baltic Nest Institute. His research deals with coastal zone biogeochemistry issues and the effects of riverine transport of biogenic elements on coastal areas.

Address: Baltic Nest Institute, Baltic Sea Centre, Stockholm University, Stockholm, Sweden.

Address: Applied Environmental Science, Stockholm University, Stockholm, Sweden.