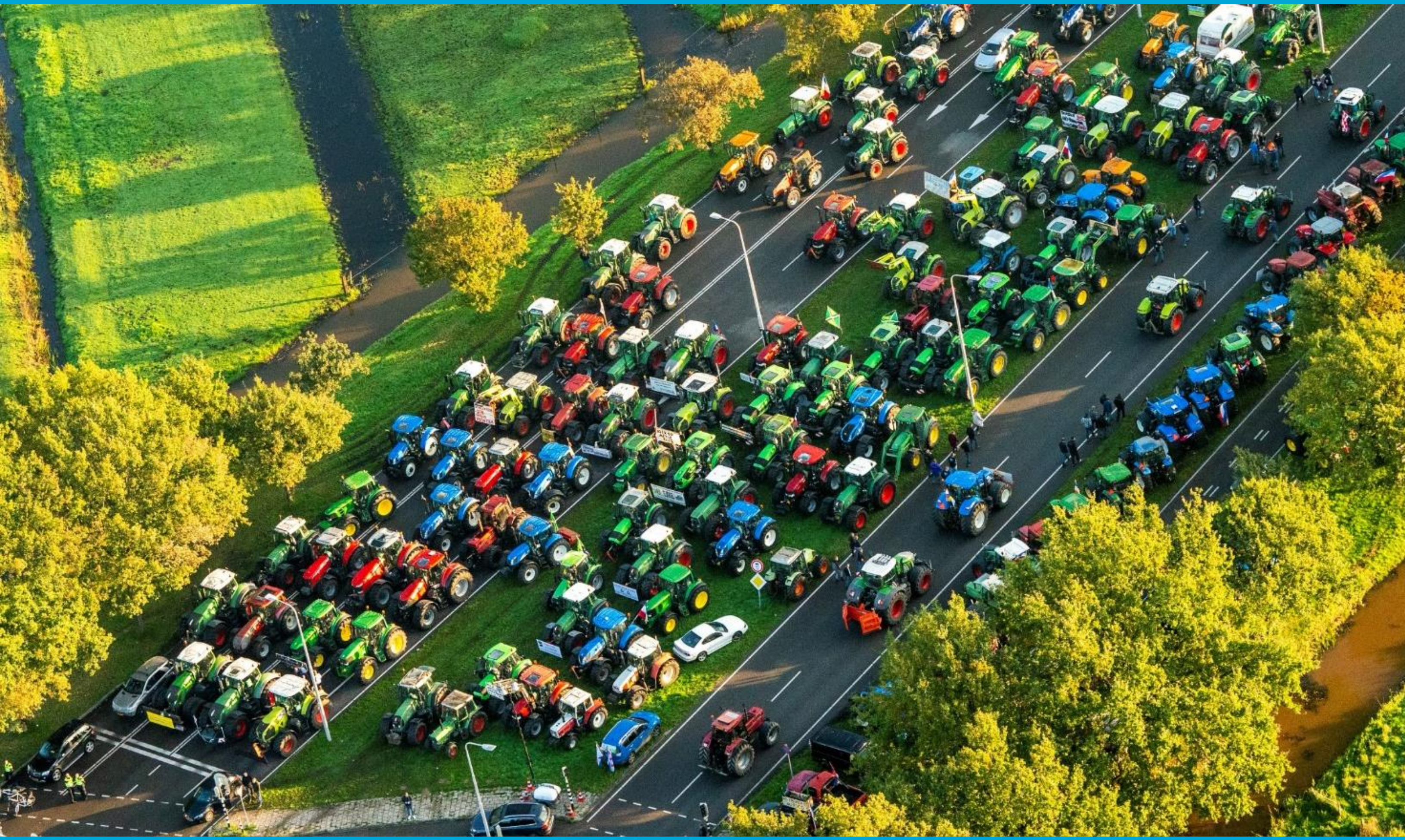
Modelling the Dutch Nitrogen Crisis

Exploring the potential of an ecological policy approach

J.M.P. Lohle



Modelling the Dutch nitrogen crisis

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By

J.M.P. Lohle

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Full name:	Jurjen Michaël Paul Lohle		
Student number:	4376781		
Date:	November 27th of 2021		
Faculty:	Technology, Policy and Management		
Program:	Engineering and Policy Analysis		
Thesis committee:	1st Supervisor:	Dr. ir. W. Auping	TU Delft
	2nd Supervisor & Chair	Prof.mr.dr. H. J. A.de Bruijn	TU Delft

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PREFACE

With this thesis I bring an end to my studies at the TU Delft. Throughout my time in Delft I have finished the TPM Bachelor, spend time at Delft Hyperloop and with this Thesis comes and end to my TU Delft journey. The writing of this thesis has been challenging, and I want to thank my close friends and family for their continual support. I wish to thank my supervisors Willem Auping and Hans de Bruijn for their enthusiasm and knowledgeable insights throughout this process. Also, I would like to thank Marcel Scheele for taking the time to familiarize me with the judicial context of the nitrogen crisis. A special thanks goes out to Thomas Jonkers, Pepijn Bouwmans, Max van Eck and Can Yildiz, for taking the time to proof read my thesis. Your contributions have helped steer this thesis towards an end result I can be proud of, and for that I am grateful.

Executive summary

In 2019, the Dutch nitrogen crisis emerged after the Dutch legislation on nitrogen pollution (PAS) was dismissed by the Dutch Raad van State. The dismissal followed an earlier ruling by the European court, and resulted in the immediate termination of permits respective to the ruling. The crisis resulted in large scale demonstration by farmers and lingering uncertainty in the Dutch business sector. To resolve the nitrogen crisis, the Dutch ambition is to reduce nitrogen emissions by 30% in 2030 and by 50% in 2035, relative to 2019. From an ecological perspective the current policy approach is limited as it focusses primarily on reducing the inflow of nitrogen to Natura 2000 areas. Policies are aimed at reducing nitrogen depositions in Natura 2000 areas below the critical deposition value (KDW). The KDW approach lacks the incorporation of complex nutrient dynamics and a link to the current state of Natura 2000 areas. Moreover, the current policy approach does not give relative importance to ecological policies measures, thereby neglecting the potential of ecological measures to compensate for nitrogen depositions.

Widening the current policy approach to better represent the ecological side of the nitrogen crisis can result in a more economically efficient outcome, with less pressure on a select group of emitters. Expansion of the policy framework requires an expansion of the current model approach, to better incorporate the ecological complexity of the nitrogen issue. To this end, not only the emissions and depositions of nitrogen, but also the nutrient stocks and the relation to the state of biodiversity must be considered. The expansion allows to investigate the potential for ecological measures to mitigate nitrogen depositions. The incorporation of ecological measures can aid the resolution of the nitrogen crisis by reducing the required amount of emission reduction to preserve Natura 2000 areas.

To explore the possible benefits of an extended policy framework, a System Dynamics model is used. The model incorporates the stocks and emission pathways of N and P, as well as an additional level of ecological complexity. To investigate the model's behaviour under circumstances of deep uncertainty an Exploratory Modelling and Analysis (EMA) approach is used. A Patient Rule Induction Method (PRIM) analysis is conducted to identify the main uncertainties underlying scenarios of low biodiversity and high costs. Low biodiversity outcomes are found dependent on the impact of nutrient pollution and the development of biomass. Compliance to EU guidelines is found to be most vulnerable to NO_x emissions from traffic. The main vulnerability for compliance to NL guidelines is related to NH_3 emissions originating from livestock. The policy analysis shows that ecological policies aimed at nutrient removal are more

effective at ensuring favorable biodiversity outcomes and low cost outcomes compared to emission reduction policies. When considering the compliance to EU emission guidelines, most scenario's show no additional required emission reduction as long as the speed limit remained at 100 km/h. The NL emission guidelines indicate a larger problem, especially in meeting the emission reduction goals for 2035. The NL emission guidelines for 2035 require a 50% reduction of livestock emissions to be reliably met.

The adaptive policy approach indicates a large potential for cost reduction through ecological compensation. By combining landscaping policies with emission reduction, over half of the reduction policies could be mitigated. For example, the combination of landscaping and livestock policy require only a 25% reduction of livestock, instead of 50%, to reliably meet NL emission guidelines. Additionally, the combination of landscaping with 50% manure reduction resulted in reliable compliance to NL guidelines which 50% manure reduction alone is unable to achieve.

The improved model outcomes in terms of cost should not be interpreted literally, as they are merely indicative of the potential for ecological measures to offset emission reduction. The potential cost savings of an ecological approach are however considerable. The model results indicate that the ecological measures are potentially able to preserve 25% of the livestock sector, which would otherwise have to be bought-out or expropriated. Considering that the Dutch agriculture sector is a multi-billion euro industry, the potential economic benefits of ecological compensation are therefore immense. In addition, preserving large parts of the industry avoids societal unrest that is likely to results from expropriation or buy-outs.

The implementation of ecological measures as a means of emission compensation is faced with scientific and legislative barriers. Before ecological measures can be put into practice as a compensatory measure for nitrogen depositions, they have to be legally accepted. Opportunities for legislative acceptance are present in article 6 of the Nature Directive. Legal acceptance does, however, require empirical substantiation which is currently lacking. To this end, efforts have to be aimed at developing the field of ecological engineering to build the required body of evidence so that emission guidelines can be offset through ecological compensation. The potential economic and societal benefits of ecological approach should be a strong incentive for policy makers and scientists to invest in research for compensatory ecological measures.

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Acronyms

N:	Nitrogen
P:	Phosphor
NO_x:	Nitric Oxides
NH_3:	Ammonia
NO_3^-:	Nitrate
NL guidelines:	Dutch emission guidelines
EU guidelines:	European emission guidelines
PAS:	The dismissed Dutch nitrogen legislation (Programma Aanpak Stikstof)

1. Introduction

1.1 The nitrogen crisis

In May 2019 the Dutch legislation for the mitigation of nitrogen (N) pollution - or PAS (Programma Aanpak Stikstof) - was dismissed by the Dutch Raad van State (Raad van State, 2019). The dismissal was based on an earlier ruling by the European Court, which deemed the PAS to be unlawful in the face of European Habitat Guidelines. The PAS legislation aimed to provide the agricultural sector with certainty, by providing permits to nitrogen emitting projects close to Natura 2000 areas (Schoukens, 2017). With the PAS, permits could be issued based on promises of expected future reductions in nitrogen emissions. The dismissal of the PAS by the Raad van State resulted in the immediate termination of permits respective to the ruling, from which the nitrogen crisis unfolded.

The nitrogen crisis brought large parts of the Dutch economy to a screeching halt. The termination of permits halted construction projects near Natura 2000 areas, and farming business expansions were stalled (Stokstad, 2019b), jeopardizing over 14 Billion worth of construction projects according to Buijs (2019), an economist at ABN AMRO. The crisis resulted in large scale demonstrations by farmers which feared for a significant reduction of their practices (Eline, 2019). In the wake of the crisis, uncertainty lingered in the agricultural, construction and business sector, as concerns remained regarding the new policies on nitrogen (Vrieselaar & Barendregt, 2021). To remove the gridlock of the economy, the Dutch government reverted to short-term policy, such as the lowering the speed limit, to reduce the amount of nitrogen emissions to legal standards (Stokstad, 2019b).

The cause of the crisis can be traced back to EU nature laws which are setup to protect ecosystems and biodiversity by regulating anthropogenic drivers for environmental change (European Commission, 2021). The preservation of ecosystems are deemed of vital importance for the functioning of earth's life support system (Costanza et al., 1997) and thereby the survival of society as they provide essential societal benefits (Mooney et al., 2009). Ecosystem functioning is heavily reliant on biodiversity as it; plays a key role at all levels of the ecosystem service hierarchy (Mace et al., 2012), provides stability to ecosystem productive capacity (Isbell et al., 2015), and plays a key role in the resilience of ecosystems functions (Gunderson, 2000; Oliver et al., 2015). The preservation of biodiversity is thus viewed as essential to ensure the long-term sustainability of ecosystems and the services they generate (Folke, 1996; Naeem, 1998). The necessity to preserve biodiversity and ecosystems resulted in the European "Biodiversity

Strategy for 2030", including the Nature and biodiversity law and the Natura 2000 network (European Commission, 2021). These policies aim to maintain and restore the habitats and species in a favorable conservation status, and avoid activities that could result in deterioration and disturbance of the habitats and species (European Commission, 2018). By providing these guidelines, European member states are obliged to meet certain goals. Member states are then free to design and enact legislation to achieve these goals (European Union, 2021).

For the preservation of biodiversity and our ecosystem, it is necessary to ensure compensation for drivers of ecosystem and biodiversity decline. According to Galloway (2008) the most significant effect that human behaviour has had on the balance of our ecosystem has been the effect on the nitrogen cycle. Nitrogen pollution is the most common form of nutrient pollution, which is a complex anthropogenic driver for environmental change (Galloway et al., 2008). Nitrogen pollution has become more persistent mainly due to the intensification of agriculture and fertilizer use, and the increase in fossil fuel combustion (Ceulemans et al., 2014). N depositions affects soils and surface water, and reduces species biodiversity mainly through processes of eutrophication and acidification (Bouwman et al., 2002; Leip et al., 2015). Increased reactive nitrogen depositions, in the form of nitrogen oxides (NO_x), nitrous oxide (N20) and ammonia (NH_3), are considered one of the major drivers for biodiversity decline throughout the ecosystem (Amon et al., 2006; R. Bobbink et al., 2010; De Schrijver et al., 2013). Nitrogen depositions are thus considered a major threat to the European Natura 2000 network (Schoukens, 2017), which is why the NL presents goals for nitrogen emission reduction in nitrogen law (Aanpak Stikstof, 2021).

For the fulfilment of the EU guidelines on the reduction of nitrogen emissions, the Dutch policies are mainly based on critical deposition loads, or KDW's (Kritische depositie waarden) (Adviescollege Stikstofproblematiek, 2020). KDW's are used to indicate if certain Natura 2000 areas are overloaded with nitrogen from areal depositions. The KDW for nitrogen is defined as the threshold above which the risk of significant quality loss of a habitat exists due to eutrophication or acidification, occurring from atmospheric nitrogen deposition (van Dobben et al., 2012). As van Dobben explains (2012), for each specific habitat a KDW value is determined based on KDW-ranges as empirically established by the UNECE, and further specified with the help of model results and expert judgement. The empirical determination of KDW's is primarily done through field or lab experiments where a N loads are artificially increased (Roland Bobbink & Hettelingh, 2010). Bobbink & Hettelingh (2010) explain that if the experiment resulted in significant changes in the ecosystem, it is inferred with confidence that N deposition were the cause. Targeted field surveys which cover gradients of N deposition are used as additional evidence to support conclusions from experimental N additions on KDW values (Roland Bobbink & Hettelingh, 2010). If empirical values are not available an average model outcome is used to determine the KDW of a habitat (H.F. van Dobben & A. van Hinsberg, 2008; van Dobben et al., 2012).

The Dutch ambition for nitrogen mitigation is to reduce nitrogen emissions by 30% in 2030 and by 50% in 2035, compared to emissions in 2019 (Adviescollege Stikstofproblematiek, 2020). If successful, 74% of the nitrogen sensitive areas will fall below the KDW norm in 2035. Additionally, measures to ensure the recovery of nature in Natura 2000 areas are to be put into legal bounds (Adviescollege Stikstofproblematiek, 2020). In the case of nitrogen pollution such measures refer to the removal of nitrogen through grazing, turfing, burning or additional mowing (Bij12, 2021).

Even though it is deemed necessary to ensure emission reduction for sustainable nature preservation (Adviescollege Stikstofproblematiek, 2020), the policy focus on KDW's is limited for a multitude of reasons. First of all, the policy focus on KDW's takes a simplistic approach to the complex issue ecological preservation, as it only focusses on the arial inflow of nitrogen into the ecosystem. The KDW perspective is limited to the arial deposition of nitrogen. KDW's are useful for assessment of semi-terrestrial ecosystems, as these ecosystems are vulnerable to nitrogen depositions through the arial emission pathway (European Environment Agency, 2010). A significant portion of the Natura 2000 areas are however aquatic (Schmedtje et al., 2011). For these aquatic ecosystems the dominant pollution pathway is not arial but aquatic, in the form of nitrate (NO_3^-) leaching (European Environment Agency, 2010). The policy focus on emission reduction by means of KDW's does therefore not guarantee sufficient preservation of these aquatic ecosystems.

Secondly, KDW's are determined mainly empirical from experiments with independent N load variations (Roland Bobbink & Hettelingh, 2010). These experiments thus do not consider interrelations with other significant factors which negatively impact biodiversity, such as phosphor (P) pollution. Similar to N, P is a major nutrient pollution and identified as the main driver for eutrophication in aquatic ecosystem (Foy, 2015; Porter et al., 2013). The impact of N and P on ecosystems are mutually dependent as they are both fundamental nutrients for the processes underlying plant growth such as; photosynthetic processes, cell growth, metabolism and protein synthesis (Chapin III et al., 2011). Additionally, the pollution of N and P is strongly correlated as the majority of pollution for both nutrients originates from agriculture practices (Ceulemans et al., 2014; CLO, 2020). Therefore, considering KDW's which are based on an independent N variation (H.F. van Dobben & A. van Hinsberg, 2008; van Dobben et al., 2012) do not give an accurate representation of the impact of N depositions on ecosystems.

A third limitation of the focus on nitrogen inflow is related to processes of acidification and eutrophication which are major drivers for biodiversity decline throughout the ecosystem (Amon et al., 2006; Roland Bobbink & Hettelingh, 2010; De Schrijver et al., 2013). These processes have an increasingly large effect on biodiversity decline based on the availability of nutrients in the soil and water (Roland Bobbink & Hettelingh, 2010). The impact of nutrients on the state of our ecosystems is thus best analysed in terms of nutrient availability, which is measured in terms of concentration. The state of the ecosystem should thus be considered based on the available stock of N and P that varies based on the inflow or outflow of nutrients. The current policy perspective lacks the integration of this ecological dynamic, as it mainly focusses on the KDW's (Adviescollege Stikstofproblematiek, 2020), and is thus limited to the inflows of nitrogen to the ecosystem. The relative importance of the N and P outflows by means of nature preservation in the form of grazing, mowing or turfing are thereby neglected. These preservation measures can create a significant outflow of nitrogen from the ecosystems, thereby compensating for a portion inflow of nitrogen and aid ecological restoration (Bullock et al., 2011). However, the current policy approach, as presented by the Dutch advice committee on nitrogen mitigation (2020), gives no clear indication on the relative importance of the proposed nature preservation measures, thereby neglecting a significant solution space.

From an ecological perspective the current policy approach thus has its limitations; as it focusses primarily on the inflow of nitrogen, lacks an incorporation of nutrient dynamics, neglects the dominant emission pathway to aquatic ecosystems and does not give relative importance to policies measures that facilitate nutrient outflows. This approach results in a limited solution space which is primarily aimed at reducing emissions, putting a disproportional pressure on a select group of emitters. Moreover, the strong focus on emission reduction cuts deep into the economic system as nitrogen emissions lay at the foundation of key economic activities such as agricultural practices, traffic, industry and construction (CLO, 2019). Widening the current policy approach to better represent the ecological complexity of nutrient in and outflows could thus result in a more economically efficient outcome with less pressure on a select group of emitters.

Currently, the models used for policy testing of nitrogen are based on emission models for air-quality such as the OPS (Sauter et al., 2020) and EMEP (Pisoni et al., 2019) models. These models exclusively model emissions and depositions through air, and do not include phosphorus pollution which occurs mostly through water (Chardon & Schoumans, 2002; Foy, 2015). The STONE model does considers both N and P and is used to test policies for manure management, but exclusively for water quality (Willems et al., 2008;

WUR, n.d.). Additionally, these models do not incorporate the stocks of N and P which are present in the soil and could affect the interaction between the N and P cycles (P. M. Vitousek et al., 2010a), and consequently the state of the ecosystem (Elser et al., 2009, 2010).

To design appropriate mitigation policies and understand the impact of N and P on the ecosystem, an integrated approach which links the stores, fluxes, and cycles of N and P is necessary (Guignard et al., 2017). Additionally, the importance of incorporating both the terrestrial and aquatic components of the system, as well as the interaction between the nutrient cycles, is critical (Grimm et al., 2003; Guenet et al., 2010; Soininen et al., 2015). Integrating the nutrient cycles of N and P in this manner can expand the knowledge on the impact of N and P on our ecosystem, which is necessary to maintain agricultural productivity whilst conserving essential biodiversity for the provision of ecosystem services (Guignard et al., 2017).

To explore the possible benefits of an extended policy framework, a model which incorporates the additional level of ecological complexity is required. Such a model can provide policy makers with insights on additional solution avenues to resolve the nitrogen issue. The broadening of the solution space can help find a more economically optimal outcome as it provides insights on the relative impact of policies regulating both the in- and outflows of nutrients from the ecosystems. The model outcomes can help open up the political discourse by providing a broader set of policy options which can be more accurately tailored to the ecological and economic needs. The broader solution space can help alleviate some of the pressure on the small group of emitters, circumventing possible resistance and public unrest originating from this group, and facilitating the finding of a broader support base for policies on nitrogen mitigation.

1.2 Political practice

The theoretical implication that a broader policy perspective would benefit the management of the nutrient pollution should also consider the political reality. A theoretical improvement can only be helpful to the real world issue, if it is accepted into the decision-making process. Multiple barriers for the acceptance of the model results exist. These barriers are rooted in either the regulatory framework or the complexity of the stakeholder environment. These barriers make up the political reality and must be taken into account when evaluating the usefulness of the results of this research.

First of all, the issue of nutrient pollution is complex from a regulatory perspective as it is impacted by both Dutch and European laws and guidelines on emissions and nature preservation. European laws for the protection of Natura 2000 areas follow from the European Nature and Habitat directive (European Commission, 2018, 2021), and result in nitrogen deposition regulations, or KDW's (Kritische Depositie

Waarden). These KDW's are further processed in the Dutch nitrogen law in terms of emission reduction for the protection of the Natura 2000 areas (Aanpak Stikstof, 2021). Moreover, the protection of Natura 2000 areas is regulated in the Dutch Nature law which prohibits any activities that can results in the deterioration of the areas (Overheid, 2015). Additionally, European guidelines for water quality for the aquatic pollution of P and N European must be followed (Schmedtje et al., 2011), as expressed in the Dutch "Kader Richtlijn water", or KRW (RIVM, 2020c). Lastly, European nitrogen emissions guidelines are setup to ensure air quality standards (European Union, 2016). These regulatory frameworks must be considered to determine the feasibility of the model findings to influence the decision-making process.

Aside from the legal feasibility, the usefulness of the model outcomes depends on the impact of the findings on the decision-making process (Figure 1). If the model findings can help broadens the solution space it could open-up the discourse in previously static parts of the decision-making process. Alternative policy routes could be explored, allowing for a distribution of mitigation efforts. Such solutions would alleviate the disproportional high pressure on a small group of stakeholders, thereby, reducing the amount conflict and resistance originating from this group. If so, the benefits to the decision-making process could out-weight the disadvantages of the efforts required for the implementation of a new policy approach. If, on the contrary, the solution space is narrowed by the findings, it will likely deepen the conflict and increase the difficulty of decision-making.

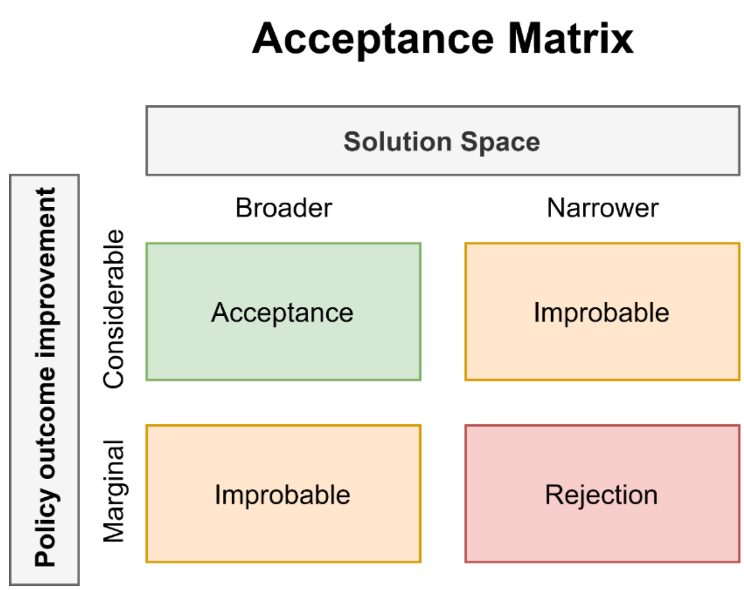

Figure 1: Acceptance matrix for model results into the political discourse

1.3 Research questions

The challenge that government policy faces, in light of the economic relevance of nutrient policies and the importance of reducing them, can be distilled into the following main research question:

" How can an expansion of the nitrogen policy framework – including N and P emissions, depositions and storages – contribute to the optimization of robust and economically efficient policies for the resolution of the nitrogen crisis? "

In formulating the answer to the research question, a number of mutually complementary sub-questions are formulated. First, a literature study is performed to explore the potential benefits of an extension of the policy scope on nitrogen pollution for the Dutch goals of nature preservation (sub question 1). Secondly, the relevant components to the complex system of policy, pollution and economic factors have to be mapped out in a conceptual model (sub question 2). Hereafter, these factors and their relations have to be correctly modelled to represent the systems behaviour and support the answering of the main research question (sub question 3). After the implementation of the model, a policy analysis must be conducted to assess the performance of policies over the entire space of uncertainty (sub question 4). Lastly, based on the results, recommendations are made for the improvement of current policies for the resolution of the nitrogen crisis (sub question 5).

Sub questions:

1. Why could an integration of the nutrient cycle in the Dutch nitrogen policy approach – including emissions, depositions and storages – be beneficial?
   - Chapter 1 – Introduction
2. How can the benefits of a new policy approach to the Dutch nitrogen crisis be analysed?
   - Chapter 2 – Methods
3. How can the Dutch policy issue of nutrient pollution - in the form of N and P - for nature preservation and economic performance be implemented in a model?
   - Chapter 3 – Model
4. Which policies for the mitigation of nutrient pollution are most robust and cost efficient for the preservation of Nature 2000 areas and the resolution of the Nitrogen crisis?
   - Chapter 4 – Results
5. In which ways can the Dutch policy approach to the nitrogen crisis be improved, following the extended scope on nutrient pollution and nature preservation?
   - Chapter 5 – Discussion & Conclusion

2. Method

For the analysis of the system of study a combination of analysis methods is presented. A System Dynamics modelling approach is presented, which allows the future exploration of the pollution of N and P to Nature 2000 areas, its impact on biodiversity, as well as the economic implications of policy measures. Moreover, the Exploratory Modelling and Analysis (EMA) tool is presented for the analysis of the uncertainties inherent to the complex components of the system of study.

2.1 System dynamics

The system of study relates to the nutrient pollution cycles of nitrogen and phosphorus. The cycles are interrelated with biodiversity, regulated by emission guidelines and determinative for policy costs. This system comprises factors of *non-linearity* (e.g., biodiversity development), *feedback loops* (e.g., N and P cycles), *accumulative processes* (e.g., soil and biomass storage of N and P) and *delay structures* (e.g., degradation and leaching processes). As defined by Lane (1999) these characteristics are central for dynamic systems.

System dynamics is a model-based approach which allows for the mathematical simulation of complex interrelations in a dynamic system (Forrester, 1961; Lane, 1999). The mathematical simulation of a system dynamics model is based on a system of coupled, nonlinear, first-order differential or integral equations (Richardson, 2019). The approach encompasses the necessary capabilities to handle factors of non-linearity, feedback loops, accumulative processes and delay structures (Lane, 1999), which are central to the system of study. A System Dynamics approach is therefore able to model the key elements relevant to the system of study.

2.2Exploratory modelling and analysis

For an effective analysis of the issue of nutrient pollution, an analysis approach is required that has the properties to deal with the aspects of deep uncertainty, inherent to the system. The characteristic of deep uncertainty follow from a lack of understanding of the inherently complex system of ecology, as well as the lack of consensus on key values by which the nitrogen crisis should be resolved. Together these factors of uncertainty create a situation of deep uncertainty (Kwakkel et al., 2010; Lempert et al., 2003).

First, an exploratory approach is used to analyse the models complex behaviour over the entire space of possible assumptions (Bankes, 1993). Additionally, an uncertainty analysis (Bryant & Lempert, 2010) is performed to identify combinations of uncertainties that are highly predictive to certain outcomes of interest. The EMA workbench provides a Patient Rule Induction Method (PRIM) tool, which can be used to this end. A policy analysis can then be conducted over the determined space of uncertainty, relevant to the problem. The policy analysis is conducted following a full-factorial design, where each policy is tested for each scenario. Subsequently, the policy results are qualitatively analysed based on graphs, that display the models behaviour and outcomes of interest.

2.3 Data

For the realization of the study, data is required for the quantification of the system factors. As the System Dynamics approach consists of many different factors and relations, large amounts of reliable input data is necessary to ensure valuable model output. It is, therefore, important that the used data is both valid and comprehensive for the entire system. Reliable public databases, such as provided by the "Compendium voor de Leefomgeving" (CLO) (CLO, 2021) and the "Centraal Plan Bureau voor de Statistiek" (CBS) (CBS, 2021), are used to acquire reliable and comprehensive input data. Additionally, data from literature is used to provide data on emission values and coefficients to model the flow of nutrients throughout the Dutch ecosystem. Sampling techniques incorporated in the EMA workbench are utilized to compensate for the inaccuracies and uncertainties in the data.

3. Model

A model is constructed for the analysis of the Dutch policy issue of nutrient pollution. The model encompasses the relevant subsystems of the policy issue related to the nutrient cycle, biodiversity, pollution sources and policies. This chapter describes the main components and outlines of the model. A more detailed description is presented in Appendix F to L.

3.1 Nutrient cycle

At the fundament of the nutrient pollution issue, lays the nutrient cycle. The nutrient cycle encompasses the flows of nutrients in, out and throughout the Dutch ecosystem. The pollution pathways of nutrients, which in this context refers to N and phosphor P, is facilitated by a multitude of mediums, namely; atmosphere, water, soils and biomass (Berhe et al., 2010). Nutrients can accumulate in these mediums but can also be transported to other mediums through a variety of processes (Figure 2). Throughout these processes N and P nutrients take a variety of forms with particular characteristics related to emission behaviour, as further explained in Appendix M and N.

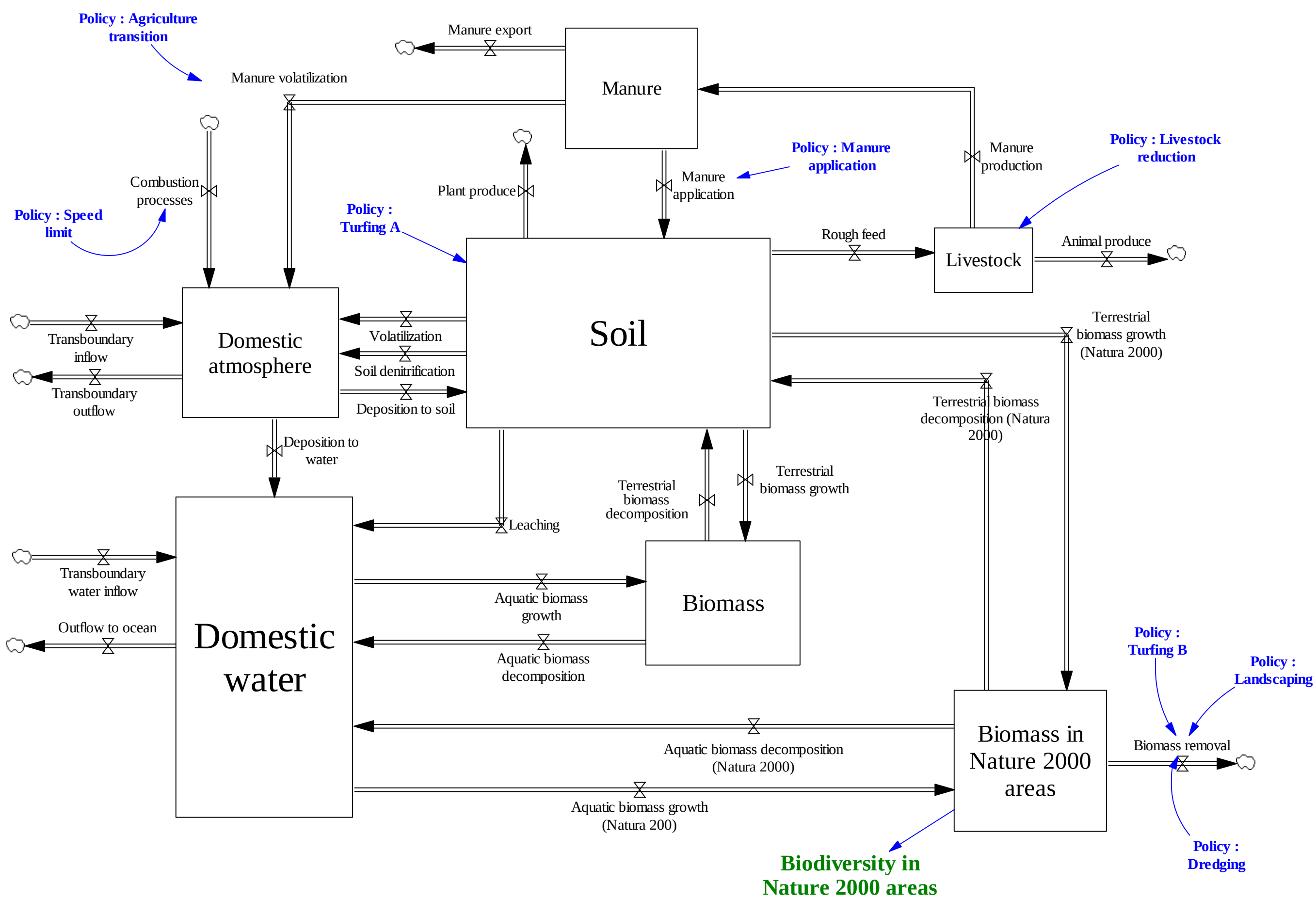

Figure 2: Conceptual overview of relevant nutrient stocks, flows and policies to the Dutch nitrogen crisis

N and P share the same emission pathways and accumulation processes throughout the soil, water and biomass mediums. An exception on the similarity of N and P cycling is the arial pathway that goes in and out of the atmosphere. In practice P can be distributed throughout the atmosphere in the form of dust. However, this pathway is strongly dependent on meteorological factors, and negligible in terms of the global phosphorus cycle (Berhe et al., 2010). Additionally, arial P emissions are characteristic by a short emission distance, making the depositions highly dependent on geographical location (Y. P. Wang et al., 2010). Since this research does not sufficiently include geographical dynamics, and the arial pathway of P is often deemed insignificant, the arial pathway of P is excluded in this research.

Medium 1 : Soils

The soil is the central medium for the flow of nutrients throughout the Dutch ecosystem for a multitude of reasons. First of all, soils are used for agricultural practices such as the cultivation of crops and the grazing of cattle. Such soils are referred to as culture grounds and are of significant economic interest (Adviescollege Stikstofproblematiek, 2020). Culture grounds are also a major source of nutrient pollution as manure and fertilizers that are applied to these soils result in NH_3 volatilization to the atmosphere (Amon et al., 2006) and result in nitrate and phosphorus leaching to waterbodies (Chardon & Schoumans, 2002). Additionally, soils provide the bases from which terrestrial biodiversity develops. Soils in terrestrial Natura 2000 areas provide the nutrients for biomass to accumulate. From the accumulation of biomass in these areas, biodiversity emerges. Hence, the state biodiversity in terrestrial Natura 2000 areas is directly linked to soils. Soils are thus central to the policy issue of nutrient pollution as they are located at the intersect of economic activity and nature preservation.

Medium 2 : Water

For this research the inland waters of the Dutch aquatic ecosystem are considered. The inland water system in the Netherlands is characterized as a river delta (Van Der Brugge et al., 2005). A river delta is defined as a water system where all the water that flows in also flows out. From this characterization and the assumption that groundwater levels remain the same, the Dutch water system can be modelled based on the inflow and outflow of water. The model for the Dutch river delta distinguishes between three waterbodies, namely; Agriculture water, Regional water and Rijkswateren. This demarcation of waterbodies is required to match available data on nutrient concentrations and waterflows (Ministerie van Volksgezondheid, 2016; RIVM, 2020c). The flow of water throughout these waterbodies facilitates the transportation of nutrients throughout the Dutch river delta. When nutrients accumulate in these waterbodies they can accumulate into biomass from which biodiversity emerges. With a significant portion of Natura 2000 areas being aquatic, water is hereby directly related to nature preservation goals (Rijkswaterstaat, 2021)

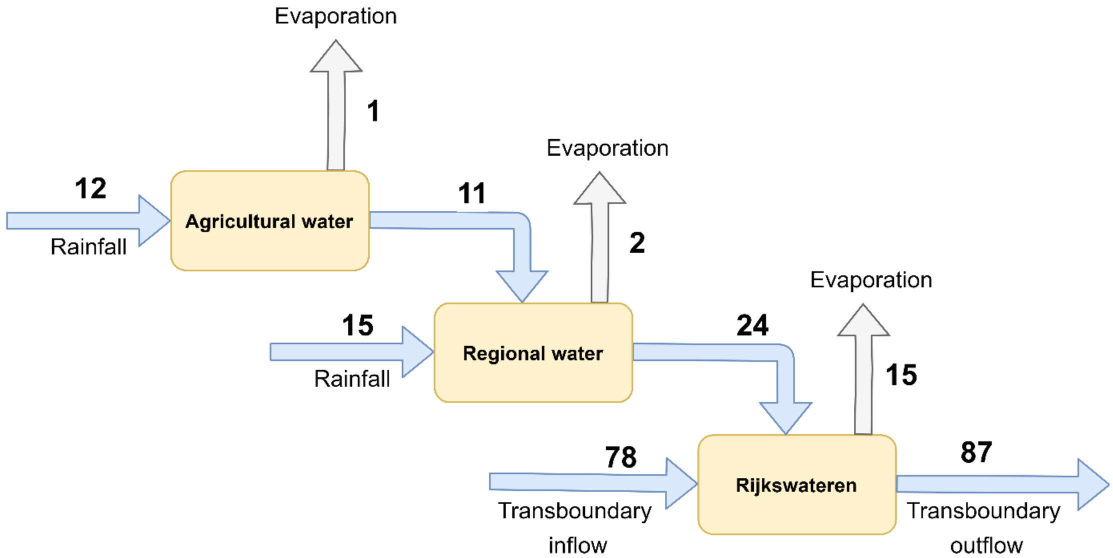

Figure 3: Conceptual overview of the flow of water through the Dutch river delta

Medium 3 : Atmosphere

The amount of reactive nitrogen (Nr) in the atmosphere is central to the Dutch system of nutrient pollution. The atmosphere is the medium through which nutrient pollutants in the form of NH_3 and nitrogen oxides (NO_x) are distributed (TNO, 2019). The amount of Nr in the atmosphere changes based on the in- and out-flows of Nr. The in-flow of pollutants stem from either national or transboundary sources. After being taken up into the atmosphere the pollutants can either deposit or be exported outside the Dutch borders.

The deposition of NO_x and NH_3 from the atmosphere occur differently due to their difference in emission pathway. NO_x has a much longer emission pathway compared to NH_3. This means that NO_x is much more likely to be deposited outside domestic borders than NH_3. The domestic deposition rates of NO_x and NH_3 are therefore modelled based on their specific deposition rates (See Appendix E). The assumption is made that the concentration of Nr in the atmosphere remains constant due to the short residency time of Nr in the atmosphere (Nair & Yu, 2020; L. Wang et al., 2019). This assumption means that the depositions of N are directly dependent on the emissions of N. An important underlying assumption of the modelling of deposition of these nutrients is that they occur uniformly over the domestic surface area. This assumption also implies that the deposition factors of domestic and foreign emissions are equal.

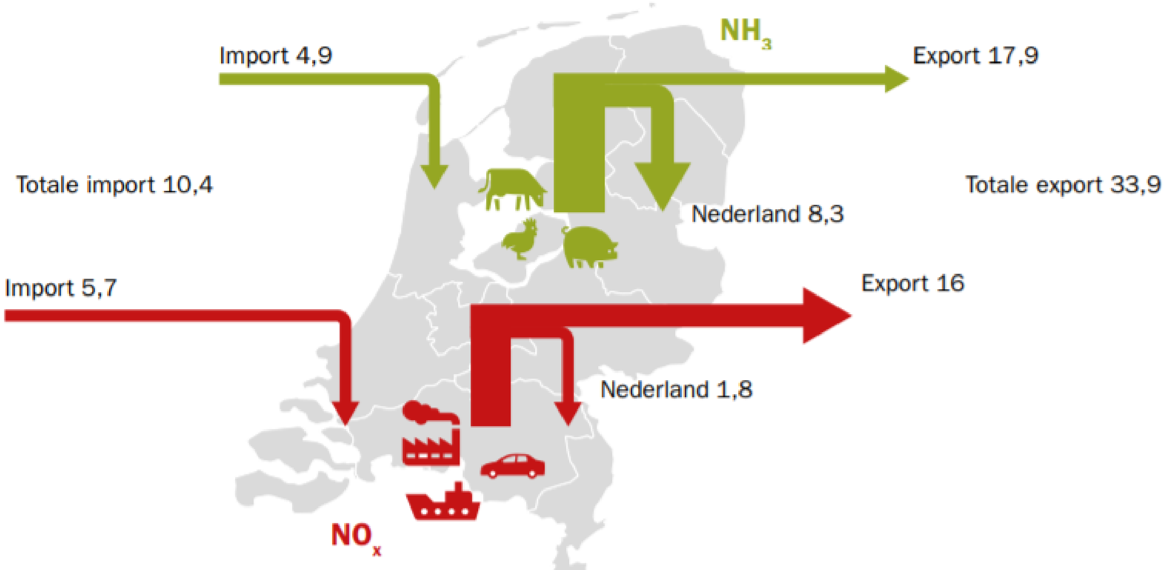

Figure 4 : The domestic and international emissions and depositions of NH_3 *and* NO_x *(source: TNO, 2019)*

Medium 4 : Biomass

The processes of biomass accumulation and decay occur in the soils and water. Nutrients accumulate in biomass, and become available again through biomass decay. The accumulation of biomass in these areas is impacted by the availability of nutrients, where high availability of nutrients results in more biomass accumulation. Moreover, the growth of biomass is impacted by the degree of biomass saturation of a habitat. A high degree of biomass saturation limits the growth of biomass in a habitat. These processes underly the development of biodiversity and are thus fundamental for the state of biodiversity in Natura 2000 areas.

Biomass accumulation also occurs in agriculture practices through the cultivation of crop, the growth of animal produce and the production of manure. The cultivation of crop is heavily impacted by the application of manure and fertilizers on culture grounds which drastically increase the availability of nutrients. In livestock, the growth of animal produce and the production of manure is directly related to the feeding of cattle (CBS, 2020b). Most of the domestic plant produce in the Netherlands is used directly as rough feed for its cattle. In turn, most of the manure production is directly used as fertilizer for the growth of crops.

3.2 N and P pollution sources

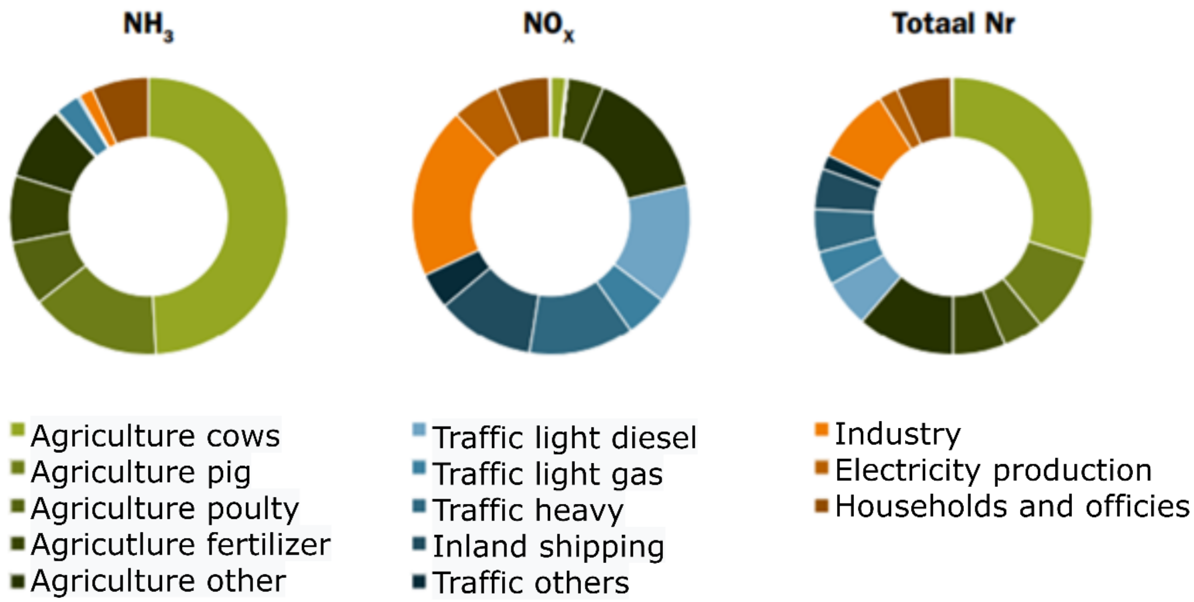

Figure 5: Domestic emission sources of NH_3*,* NO_x *and total Nr (source: TNO, 2019)*

Sources of N and P pollution are numerable, with the major sources being agriculture, traffic and industry (TNO, 2019). The pollution sources that emit to air are rooted in combustion processes (NO_x) and agriculture practices (NH_3). The major source of NO_x from combustion processes is traffic, followed by industry (Figure 5). Agriculture practices related to the cultivation of crops are the main source of nutrient pollution of water through leaching to surface water (Ministerie van Volksgezondheid, 2016). The pollution of water occurs through leaching processes of N and P resulting from manure application (Figure 6 – red). Industry and sewage and treatment plants are also major polluters of surface water.

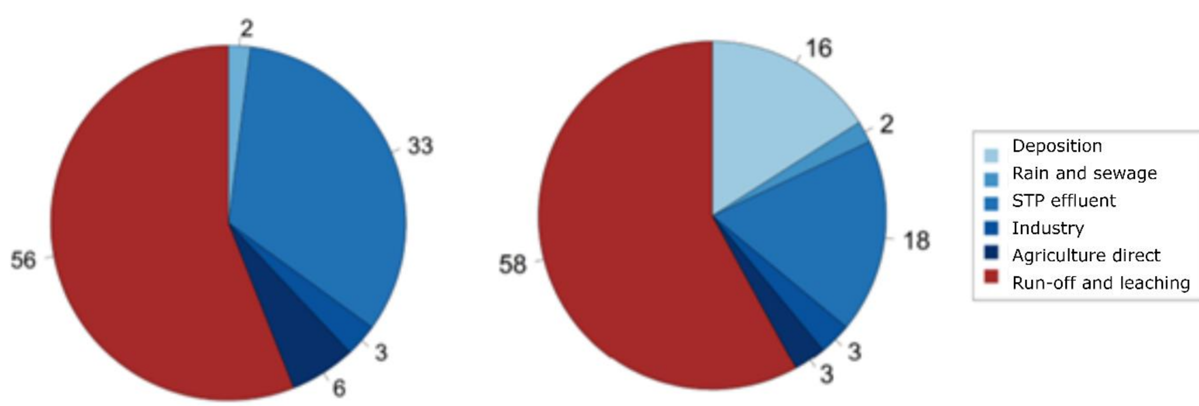

Figure 6: Domestic N and P pollution sources to surface water (2012 – 2014) (source: Ministerie van Volksgezondheid, 2016)

The emissions of N and P are regulated by guidelines such as the NEC (European Union, 2016), Nitrogen law (Aanpak Stikstof, 2021) and KRW (Ministerie van Volksgezondheid, 2016). These regulations are setup to mitigate the pollution of N and P. The NEC guidelines are setup to regulate domestic N emissions to ensure air quality standard for human health (European Union, 2016). Additional emission guidelines setup by the Dutch government in the nitrogen law are to meet nature preservation goals (European Commission, 2018, 2021). By reducing the domestic emissions, these NL guidelines aim to lower N depositions in Natura 2000 habitats to below their respective KDW's. Lastly, for the regulation of pollution to water the KRW is setup to work towards favourable water conditions (Ministerie van Volksgezondheid, 2016).

3.3 Biodiversity

Biodiversity is a key component of the Dutch nitrogen crisis, since it must be protected to meet European preservation guidelines (European Commission, 2021). These preservation efforts have led to nitrogen emission guidelines which gave way to the nitrogen crisis. Biodiversity refers to the variety of biota in a habitat, which encompasses both plants and animals (Purvis & Hector, 2000). Here, a preferential state of biodiversity refers to a habitat which holds a wide variety of biota. In most habitats plant species are adapted to nutrient-poor conditions, so they are only able to compete successfully on soils with low N levels (Bouwman et al., 2002). Nutrient pollution is a threat to biodiversity as it increases the availability of nutrients. The increased nutrient availability results in local extinction via dominance of a few competitive species, resulting in a reduction of biodiversity (Chapin III et al., 2000; Erisman et al., 2008; Lambers et al., 2011). Aquatic ecosystems are generally characterized as being sensitive to P pollution due to P limitation (Djodjic et al., 2004). Whilst terrestrial ecosystems are generally sensitive to N pollution due to N limitation (P. Vitousek & Field, 2001).. Sensitivity here indicates that an increase in the respective nutrient reduces biodiversity by giving a competitive advantage to certain types of undesirable biota.

The state of biodiversity thus emerges from biomass accumulation processes, where the distribution of biomass accumulation over the types of biota ultimately determines the state of biodiversity. The measure for the state of biodiversity is deduced from the distribution of biomass in Natura 2000 areas (See Appendix H). From this distribution the measure for biodiversity is modelled based on the share of desirable plant biomass in the total plant biomass.

3.4 Policy

In mitigating the issue of nutrient pollution, the government has certain policy measures at its disposal. Currently, the main focus on the Dutch government is aimed at reducing emissions by imposing policies to lower the speed limit, to improve livestock stables, reduce livestock, improve manure handling and regulate fertilizer and manure application (Adviescollege Stikstofproblematiek, 2020; RIVM, n.d.). Additionally, the political narrative for emission reduction has been focussed at reducing the livestock (Kuiper & Rutten, 2021). The reduction of livestock could be a solution for the nitrogen crisis, as it would bring Natura 2000 areas below the KDW's (Adviescollege Stikstofproblematiek, 2020). For the policy analysis, the reduction of livestock is taken as the default policy alternative to reduce emissions to mandatory levels, in case guidelines aren't met. The cost related to livestock reduction are therefore used as a cost measure for the inability of policies to meet the NL or EU emission guidelines.

An alternative policy approach to the one aimed at emission reduction exists and is instead aimed at ecological improvement. The central issue of the nitrogen crisis is related to the state of biodiversity in Natura 2000 areas, which has to be protected from nitrogen depositions (European Commission, 2021). If ecological measures are successful at making Natura 2000 areas more resilient to nutrient pollution, this could result in lower requirements for emission reduction. Consequently, such measures could mitigate the pressure on nitrogen emitters and limit the ramifications of the nitrogen crisis.

Currently, Natura 2000 areas are already managed for preservation and improvement of biodiversity (European Commission, 2018). The European commission obliges member states to report the efforts to preserve and improve biodiversity in a management plan, which includes; the goals, measures and responsibilities for the execution of the plan. The ecological policy measures proposed here are thus an addition to the basic management plants of Natura 2000 areas. The following ecological measures are included in this research: *Landscaping*, *Turfing* and *Dredging*. Landscaping is aimed at removing undesirable plants from Natura 2000 areas. By removing undesirable plants from the habitat, it allows for the improvement of biodiversity. Turfing refers to the removal of the top layer of the soil. Turfing thereby removes all the biota on the soil but also the excess nutrients that reside in the top soil. Lastly, the ecological measure of dredging is considered. dredging. Dredging is similar to turfing but for aquatic ecosystems and removes the nutrient rich slip from trenches or streams.

3.5 Validation & verification

To determine the correctness and usefulness of the model, verification and validation of the model is required. Verification tests are done to ensure that the model is working properly. Validation of the model is used to determine the usefulness of the model to represent the real world system.

3.5.1 Verification

Model verification occured ccontinually throughout the model construction process. The stocks and flows that were incorporated in the model were based on real world data as much as possible. Since the model is simulate from 2000 to 2050, historic data is available to verify the model values. Moreover, a parameter verification and unit check is conducted. The unit check conducted with the built-in Vensim tool resulted in no errors, which is a clear indication that the units and parameter relations are correctly setup. Moreover, emission trajectories were verified to match with historic data from 2000 to 2020, and follow expected trajectories (Figure 7 and 8).

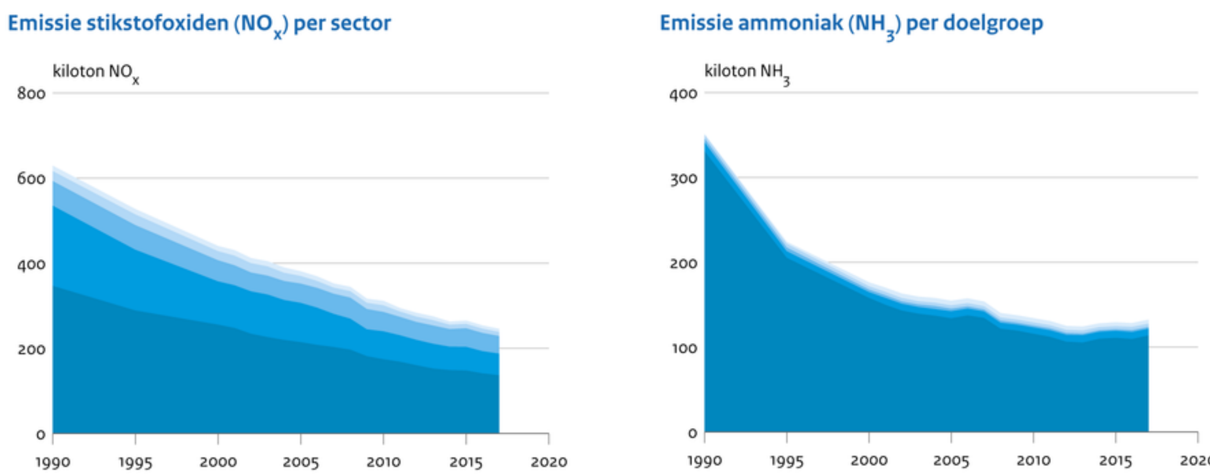

Figure 7: Domestic NO_x and NH_3 emission data (source: (RIVM, 2019)

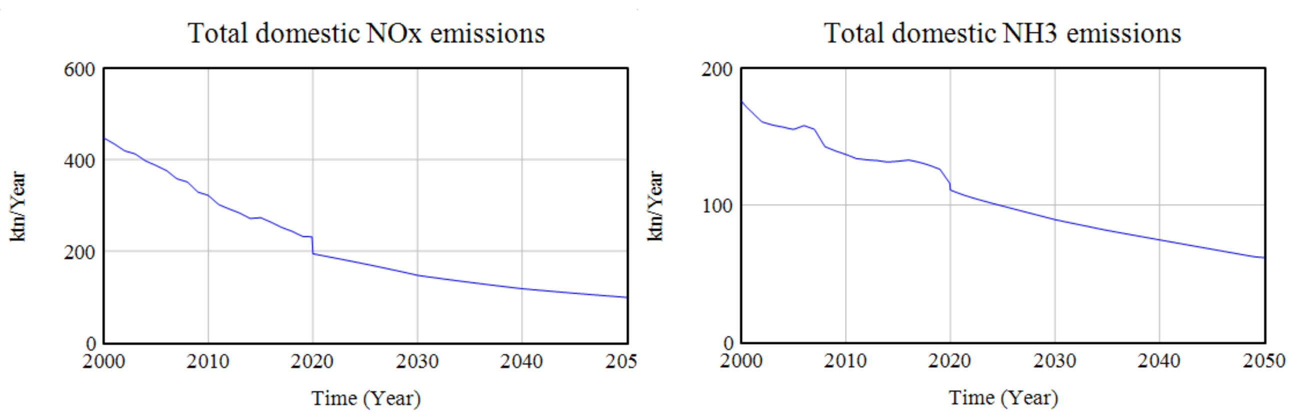

Figure 8: Model results on domestic NO_x and NH_3 emissions

3.5.2 Validation

The model validity is determined based on its usefulness to address the real world issue of nitrogen pollution. To this end, the usefulness of the model to represent critical parts of the nitrogen crisis is discussed in this section.

Compliance to emission guidelines

The model is accurately able to represent is the emissions resulting from domestic sources (Figure 7 and 8). This characteristic allows for a detailed analysis of the ability of policies to meet NL and EU emission guidelines. What the model lacks is the ability to represent geographical depositions. It can therefore not check if N depositions can be kept below the KDW's of Natura 2000 areas. However, emission guidelines are setup based on KDW's in terms of national emission reduction goals (Adviescollege Stikstofproblematiek, 2020). The nitrogen crisis is in large a consequence of these emission guidelines. So, even though the model cannot analyse the KDW compliance directly, it is still useful for an investigation of the nitrogen crisis in terms of compliance to emission guidelines.

Nutrient cycle

The complexity related to the nutrient cycle and emission pathways was significantly reduced in this research. The research for instance does not incorporate differences in soil type (e.g. clay, sand, loam etc.). This simplification neglects the difference in biota, nutrient concentration and leaching processes for each type of soil. Similarly, simplifications were made for the cycle of nutrients through waterbodies. Moreover, simplifications were made regarding the arial distribution of nitrogen. Assumptions regarding the uniform distribution of nitrogen depositions disregard the non-linear nature of nitrogen emissions and depositions. The assumption neglects the geographically dependent impact of nitrogen depositions on Natura 2000 areas.

The model simplifications of the nutrient cycle only allow for an analysis of the overall flow of nutrients. It lacks details related to geography, meteorology, soil specific pollution and biota characteristics. These simplifications make it impossible to analyse regional specific policies for the mitigation of the nitrogen crisis. The goal of this research is, however, not to design regional specific policies, but instead provide a proof of principle for alternative policy avenues. These model limitations therefore do not reject the validity of the model. The model limitations do, however, indicate that the model should be further developed to facilitate actual policy design. To this end, a more detailed modelling approach is required that accurately describes the relation between the nutrient cycle and the state of ecology, by including the factors of geography, meteorology, soil specific pollution and biota characteristics.

Ecological approach

Ecology is an extremely complex systems which had to be drastically simplified for this research. Ecology is modelled to represent biodiversity in Natura 2000 areas, and follows from the flow of nutrients and the resulting assimilation and decomposition of biomass (Paragraph 3.1 & 3.3). The main simplification of the ecological approach relate to the state of biodiversity and the types of Natura 2000 areas. Firstly, the state of biodiversity is determined based on the ratio of desirable and undesirable plants. Secondly, no distinction is made between the types of Natura 2000 areas. The validity of the ecological approach is hard to confirm due to a lack of data on the state and development of biodiversity in the Netherlands.

What is known, is that many areas are in an unfavourable status (Adviescollege Stikstofproblematiek, 2020), but this does not allow for a validation of this research's biodiversity results. The model does, however, allow the comparison of the effectiveness of policies to influence biodiversity. The results of the model can thus be used to investigate the relative effectiveness of policies is to improve the state of biodiversity over time. Moreover, the model results can be used to identify general trends of biodiversity development. The results thus allows for an initial exploration of alternative policy approaches to the nitrogen crisis. As a proof of principle, the simplistic ecological approach is therefore deemed sufficient.

Economic dimension

The economic dimension of this research is represented by the total policy costs that are calculated for each policy approach. The economic dimension is hereby reduced to a simple cost approach, and neglects the deeper economic impact of each policy approach. This means that the economic dimension of this research only allows for a comparison between policies on an investment cost bases. As an initial exploration of the policy options, this approach is deemed sufficient. However, to fully grasp the economic impact of mitigating the nitrogen crisis, a more comprehensive economic approach is required.

3.6 Experimental setup

The computer simulation program Vensim (Ventana System, 2010) is used for the implementation of the system dynamics model. The experiments are conducted over a time span of 50 years, ranging from the year 2000 to 2050. The timeframe was set to verify model behaviour for the first 20 years and allow for policy analysis and ecological performance for the 30 years hereafter. The timeframe for policy analysis (2020 – 2050) hereby covers the emission goals that are set for 2030 and 2035, and provides long term behavioural insight. The simulations were conducted with the smallest possible timestep in Vensim (Timestep = 0.03125), for maximum accuracy.

The model is analysed with the Exploratory Modelling Workbench (EMA-workbench) in Python. A Latin hypercube sampling approach is used to sample scenarios from the space of uncertainties. The sampling method uses a random uniform sampling method. Sampling is done based on the maximum and minimum values of uncertainties, which define the uncertainty space (See appendix O). The result is a uniform distribution of scenarios over the uncertainty space.

The analysis of the systems behaviour is firstly done through behavioural experiments which explore the models behaviour without policies. Through uncertainty analyses with the PRIM tool the behaviour of model outcomes is explored through identification of groups of uncertainties that explain outcomes of interest. Policies are then tested over a selection of the uncertainty space, which is relevant to the policy approach whilst still being representable for the entire space of uncertainties. First policies for ecological improvement are tested. Based on these results, the adaptive policy approach is explored. The adaptive policy approach considers a policy framework which allows for the loosening of emission guidelines based on the biodiversity performance of policies.

4 Results

In this chapter the results from the model analysis are presented and discussed. First the model behaviour is presented without the influence of active policies. From here, uncertainty analyses are conducted to investigate which combinations of uncertainties explain the model behaviours of interest. Hereafter, the effectiveness of policies are tested. The effectiveness of policies are investigated from three angles, namely; ecological, European emission guidelines (EU guidelines) and Dutch emission guidelines (NL guidelines). Here the NL guidelines approach incorporates the adaptive policy where NL guidelines are made dependent on ecological performance of Natura 2000 areas.

4.1 Model behaviour

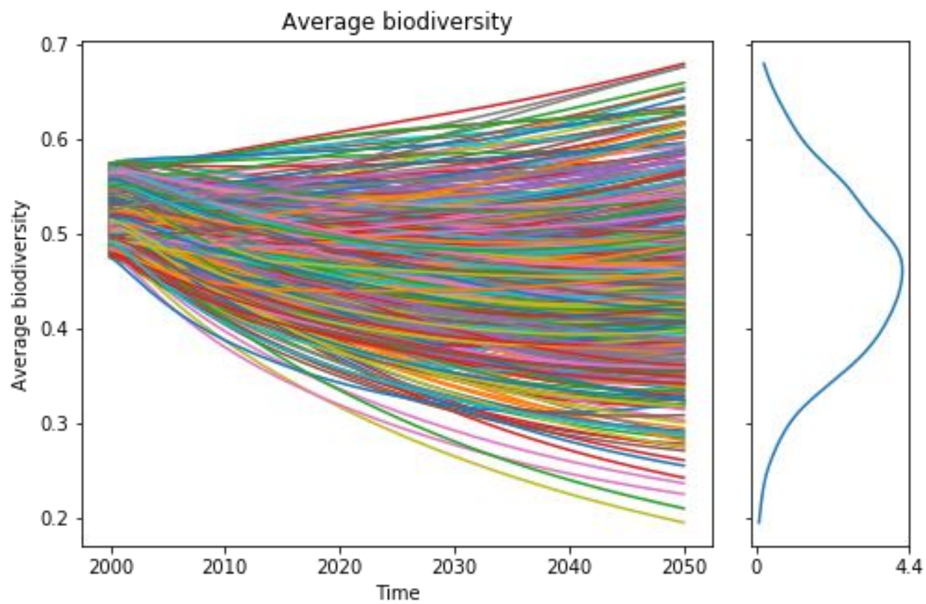

Figure 9 : Behaviour of average biodiversity scores for different scenarios without active policies

The model shows a strong variability in the outcomes of biodiversity (Figure 9). The average biodiversity scores covers both aquatic and terrestrial biodiversity performance (See Appendix H). The model outcomes of terrestrial and aquatic biodiversity show different behavioural patterns (Figure 10). Aquatic biodiversity tends to diverge either up or down in relatively linearly fashion, whilst terrestrial biodiversity develops more non-linearly. In general, biodiversity in terrestrial and aquatic ecosystems is inclined to decrease over time, as can be deduced from the graphs density functions (Figure 10). These findings correspond with expectations of degenerative biodiversity performance in Natura 2000 areas. The majority of scenarios remain relatively stable with an average biodiversity score in 2050 between 0.5 and 0.4. This behaviour indicates that the average model behaviour shows a slow and steady decline of biodiversity between the year 2000 and 2050.

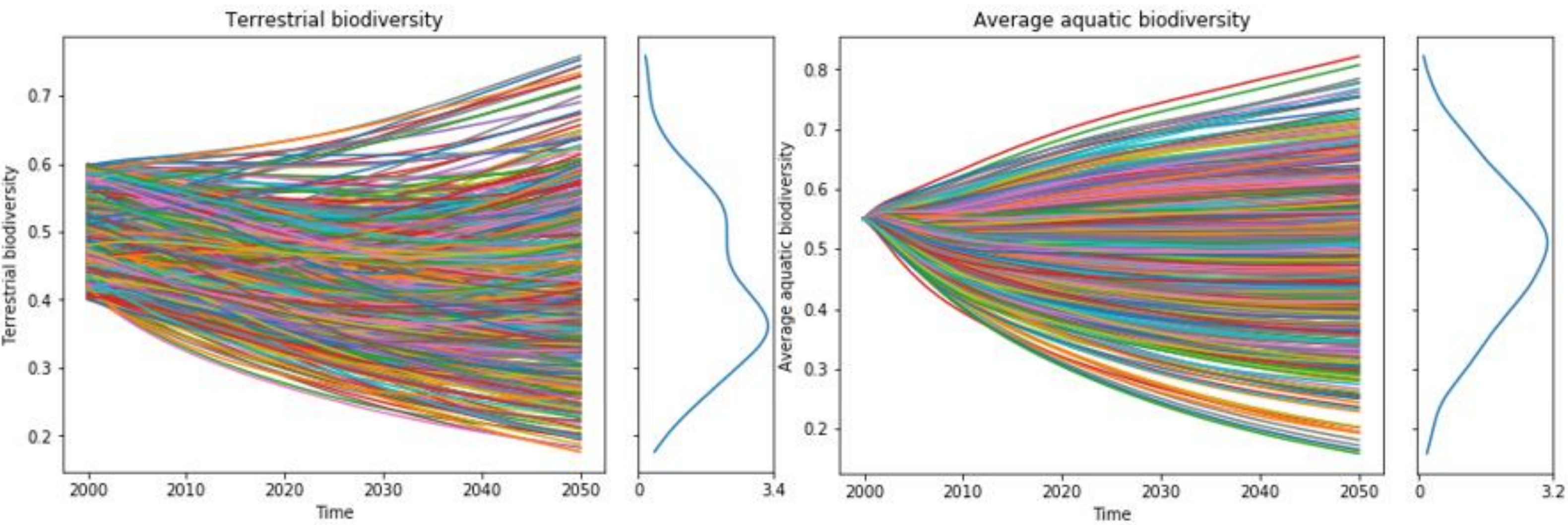

Figure 10: Behavioural comparison of terrestrial and aquatic biodiversity without active policies

In terms of policy costs related to EU and NL emission guidelines, the model shows a high variability in model outcomes (Figure 11). This variability can be explained by the uncertainties related to the emissions of NO_x and NH_3. The variability of emissions results in scenario's where the emission guidelines are not met. Livestock reduction is used as a default option to bring emission down to the required levels, resulting in policy costs. Interestingly enough, the EU guidelines result in significantly lower policy costs than the NL guidelines (Figure 11). From the density function of the EU graph can be deduced that in most cases the EU guidelines are met without the use of additional policy. The NL guidelines, however, always result in additional required emission reduction. Especially the emission goals for 2035 show to be a problem as they result in a huge leap in policy costs over the entire space of uncertainties.

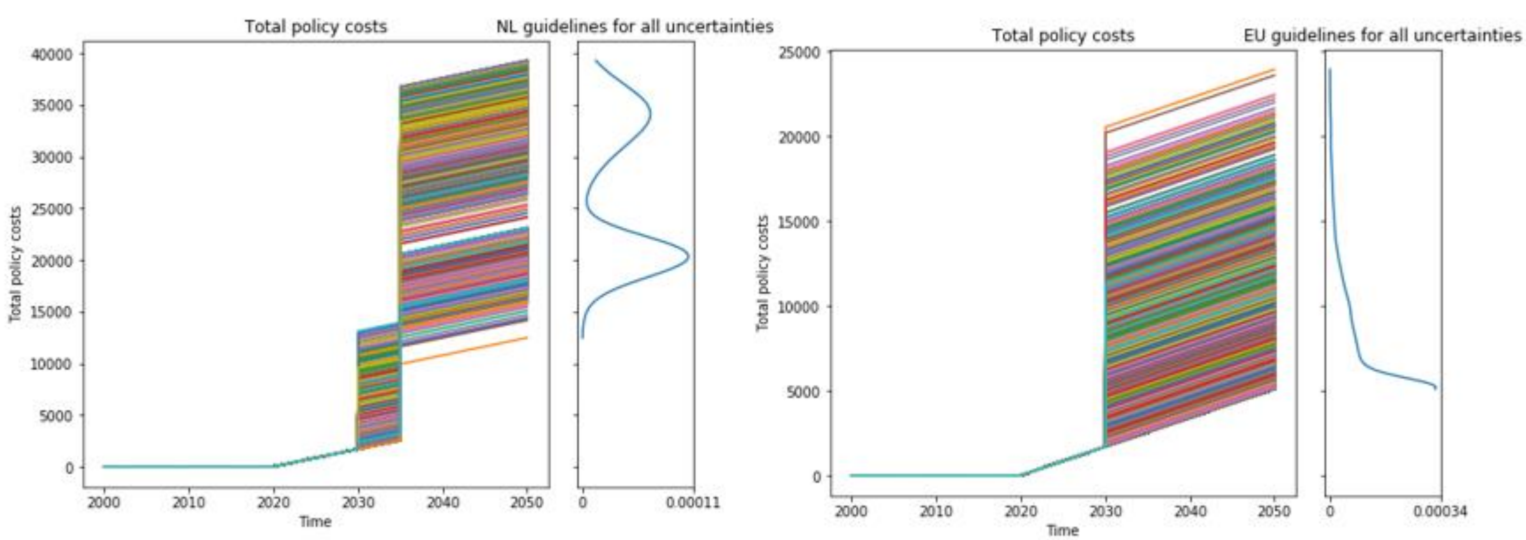

Figure 11: Policy costs comparison for NL and EU guidelines, without active policies

4.2 Uncertainty analyses

Uncertainty analyses are conducted to identify the dependency of model behaviour on independent variables. The values attributed to these variables are subject to parametric uncertainty (See Appendix O). The uncertainty related to these values influence the behaviour of the model. The outcomes related to the uncertainties represent scenarios. In order for policies to be robust, they must result in desirable outcomes over the entire space of uncertainties. Meaning, even in extreme cases of negative performance, such as low biodiversity scores or high emissions, policies must induce favourable outcomes. Identifying such vulnerabilities are critical for the design of robust policies. An uncertainty analysis is thus conducted to identify the vulnerabilities of the system which result in worst case scenarios.

To investigate the robust policy avenues, three uncertainty analysis are conducted. First, an analysis on the ecological behaviour is carried out, to analyse the uncertainties that underly scenarios that result in undesirable biodiversity outcomes. Hereafter, the uncertainties underlying the compliance to EU and NL guidelines are investigated. Conforming to EU guidelines on nitrogen emissions is crucial as these guidelines are binding (European Union, 2016). Similarly, NL guidelines must be met for the protection of Natura 2000 areas (Aanpak Stikstof, 2021). For both the EU and NL guidelines the measure for policy costs is used as to identify worst case scenarios.

4.2.1 Uncertainty of ecological performance

The scenarios which result in unfavourable biodiversity outcomes are investigated since these scenarios are most relevant for the protection of Natura 2000 areas. First a selection of uncertainties is made that represent the relevant space of uncertainties for the development of biodiversity (Appendix A1.1). The uncertainty selection is tested to ensure that similar model behaviour is generated compared to the entire space of uncertainty (See Appendix A1.2). As displayed in Appendix A1.3, the uncertainty selection results in comparable model behaviour for *Average biodiversity, Average aquatic biodiversity* and *Terrestrial biodiversity.* These findings indicate that the relevant uncertainties underlying the model behaviour of biodiversity are included in the uncertainty selection.

Scenarios of low overall biodiversity performance in the Netherlands are shown to strongly relate to the impact of nutrient pollution and the development of biomass (See Appendix A1.3). Terrestrial biodiversity is found to be most vulnerable to the development of nitrophilic biomass and nutrient pollution. This can be deduced from the findings that the vast majority of low terrestrial biodiversity outcomes are related to the *Initial share of nitrophilic biomass,* the *Nitrogen impact correction factor* and the *initial N concentration of the soil.* These factors indicate that terrestrial biodiversity is likely to degenerate strongly if Natura 2000 areas are highly sensitive to nitrogen pollution. This problem is exacerbated when nitrogen availability in in the soil is higher, and when more nitrophilic plants are present in the area.

The aquatic biodiversity shows a similar vulnerability, but then to phosphor pollution (See Appendix A1.3). The results show that if ecosystems are vulnerable to phosphorus pollution, aquatic biodiversity is likely to degenerate. Moreover, the development of biomass in aquatic ecosystems is strongly predictive for undesirable biodiversity outcomes. If the lifetime of phosphoric biomass is relatively long whilst the lifetime of other favourable aquatic biomass is relatively low, the state of biodiversity degenerates.

4.2.2 Uncertainty of meeting emission guidelines

For the uncertainty analysis related to policy costs for conforming to the EU and NL emissions guidelines a selection of uncertainties is made that is representable for the entire space of uncertainty (See Appendix A 2.1 & A3.1). The model shows similar behaviour and outcomes for the uncertainty selection compared to the entire space of uncertainties (See Appendix A 2.2 & 3.2). This indicates the uncertainty selection includes the relevant uncertainties that are explanatory for the policy costs related to the EU and NL emission guidelines. The selection is focused on uncertainties related to nitrogen emissions and the state of terrestrial biodiversity. The uncertainties related to emissions are extremely relevant In this context as they directly determine the degree to which the EU and NL emission guidelines are met.

EU guidelines

Uncertainties related to the cost scenario and emissions from traffic are most predictive for worst case costs outcomes in the context of EU guidelines (See Appendix A2.3). High policy costs are a measure for the degree of additional mitigation measures that are required to meet the EU guidelines. Consequently, it logically follows that the variable *high cost scenario* underlies the worst case cost scenarios for meeting EU guidelines. The rest of the uncertainties that are predictive for high cost scenarios are related to traffic (See Appendix A2.3). These findings indicate that NO_x emissions resulting from traffic are a major contributing factor in conforming to EU guidelines. The uncertainties related to traffic are twofold, with the first being the average driving distance of gas cars. When the average driving distance increases, EU guidelines are more easily violated. Secondly, the speed of the transition to electric cars is a significant predictor for EU guidelines violations. In case the transition happens to slow, due to relatively longer car lifetimes, or ineffectiveness of Dutch transition policies, the required policy costs to meet EU guidelines develop into worst case scenarios.

NL guidelines

The cluster of uncertainties that together are most predictive for high policy costs related to NL guidelines are related to the cost scenario, livestock and traffic (See Appendix A3.3). It is expected that a high cost scenario for emission reduction results in worst case cost scenarios. More interesting are the uncertainties related to livestock. Here, the uncertainty related to manure production from livestock and manure volatilization are found to be significantly predictive for worst case cost scenarios. These findings indicate that uncertainties related to NH_3 emissions from manure are a major source of concern when trying to meet NL guidelines. Especially cow cattle is central to this issue of NH_3 emissions. When trying to meet NL guidelines, the NH_3 emissions from manure produced by cattle livestock is therefore the key vulnerability of the systems compliance to NL emission guidelines.

Aside from livestock, uncertainties related to traffic are also found to be significantly predictive for worst case cost outcomes (See Appendix A). Traffic is found to be less significantly correlated to high policy costs than the factors related to NH_3 emissions. Still, the findings that car lifetime and the effectiveness of Dutch electric car policy are determining factors for worst case cost scenarios should be considered. Similar to EU guidelines, these findings indicate that the transition away from gas cars towards electric cars is vital for meeting guidelines. If the transition does not occur rapidly enough, it could result in worst case costs outcomes.

4.3 Policy analysis

In this chapter the results of the policy analysis are presented. The policies discussed in paragraph 3.4 are tested for their performance for three policy approaches. Both the effectiveness of single policies and combined policies are tested. Table 1 and 2 display the single and combined policies that are tested in this research. Based on the relevance to each policy approach, a selection of combined policies is tested. The first policy approach is the ecological approach, which tests the effectiveness of policies to enhance the state of biodiversity. Hereafter, a regulatory policy approach is taken to test the ability of policies to comply to EU and NL emission guidelines. Lastly, the policies are tested within an adaptive policy framework, in which the strictness of the NL emission guidelines is dependent on the state of biodiversity. Within each policy framework the individual performance of policies is tested, as well as promising combinations of policies. The policies are tested based on the uncertainty selection as determined in paragraph 4.2.

Table 1: Overview of single policies that are implemented in the policy analysis

Single policies	**Clarification**
No policy	This policy encompass a 100 km/h speed limit and a 170 mln/year investments in agriculture transition. These are the base policies enacted by the Dutch government.
Agriculture transition	A 340 mln/year investments in agriculture transition, instead of the base rate of 170 mln/year (Appendix L)
Dredging	A 200 mln/year investment in dredging activities (Appendix L)
Landscaping	A 200 mln/year investments in landscaping activities (Appendix L)
Livestock	A 50% reduction of livestock (Appendix L)
Livestock 25%	A 25% reduction of livestock (Appendix L)
Manure	A 50% reduction of manure application (Appendix L)
Manure 25%	A 25% reduction of manure application (Appendix L)
Turfing	A 200 mln/year investments in turfing activities (Appendix L)
Speed limit 80	A speed limit reduction to 80 km/h (Appendix L)
Speed limit 130	A speed limit increase to 130 km/h (Appendix L)

Table 2: Overview of the combined policies that are implemented in the policy analysis

Combined policies	**Clarification**
Max emission reduction	Combination of Livestock (50%), Manure (50%) and Agriculture transition policies.
Max nutrient removal	Combination of Turfing, Landscaping and Dredging policies
Land Live	Combination of Landscaping and Livestock (50%)
Land Live 25%	Combination of Landscaping and Livestock (25%)
Livestock 25%	25% livestock reduction
Livestock 130	Combination of Livestock (50%) and speed limit 130
Land Man	Combination of Landscaping and Manure (50%)
Land Man 25%	Combination of Landscaping and Manure (25%)
Manure 25%	25% Manure reduction
Manure 130	Combination of Manure (50%) and speed limit 130

4.3.1 Ecological policy approach

For the ecological policy approach, the performance of individual and combined policies are tested based on their biodiversity improvement. Within the ecological policy approach the aim is to identify which policy measures are effective in mitigating nutrient pollution with the aim of improving biodiversity. To this end, the biodiversity performance of the policies are tested based on the models biodiversity scores. Additionally, the cost of policies is analysed for a cost comparison of the policies.

Biodiversity performance of policies

Figure 12 displays the overall behaviour of biodiversity for each of the individual policy measures. The average biodiversity is a measure for biodiversity over the entire Natura 2000 network. From figure 12 it is apparent that landscaping is the policy which shows the clearest improvement of biodiversity after implementation. Biodiversity seems to reliably improve over the entire space of uncertainty after the implementation of landscaping. After L*andscaping*, *Turfing* is the only policy which shows a clear performance improvement over the other policies. *Dredging* shows a slight improvement from the norm, but can hardly be deemed significant. The other policies (e.g. *agriculture transition, livestock, manure* and *speed limit)* do not outperform the *no policy* option.

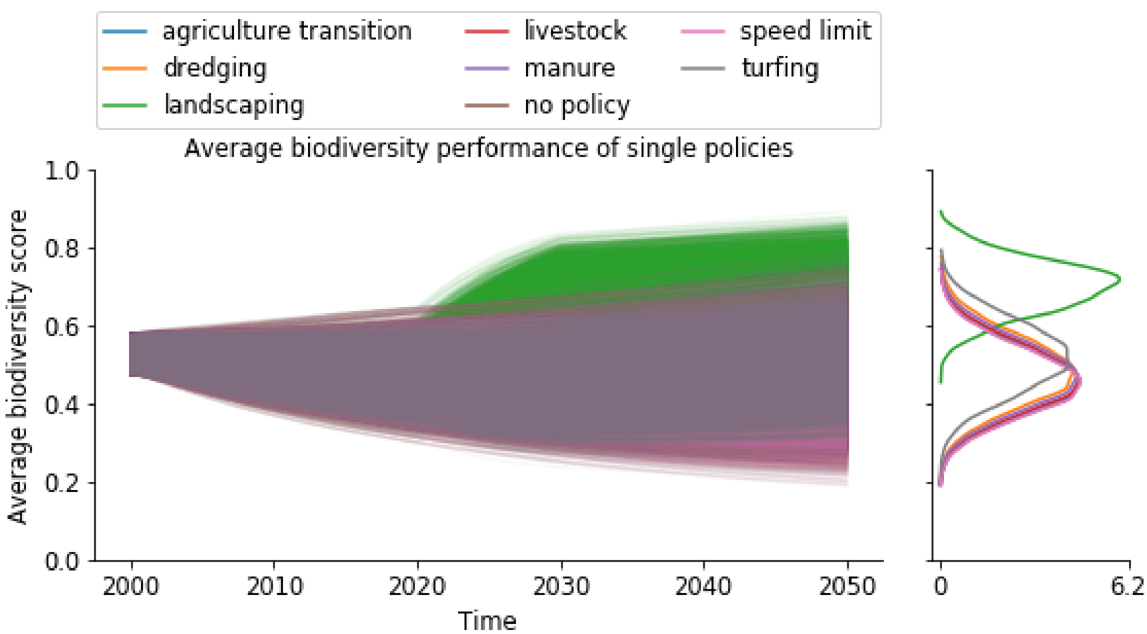

Figure 12: Average biodiversity scores for individual policy measures.

These findings are further substantiated when considering the terrestrial and aquatic biodiversity scores of policy measures (See Appendix B). Here, terrestrial biodiversity only seems to be impacted by landscaping and turfing. Moreover, aquatic biodiversity shows improvement after the implementation of

Dredging. Also, a slight shift in aquatic biodiversity performance is observable for manure application. However, the *Dredging* and *Manure application* reduction measures do not reliably improve aquatic biodiversity over the entire space of uncertainty. Indicating that they are not effective at improving aquatic biodiversity.

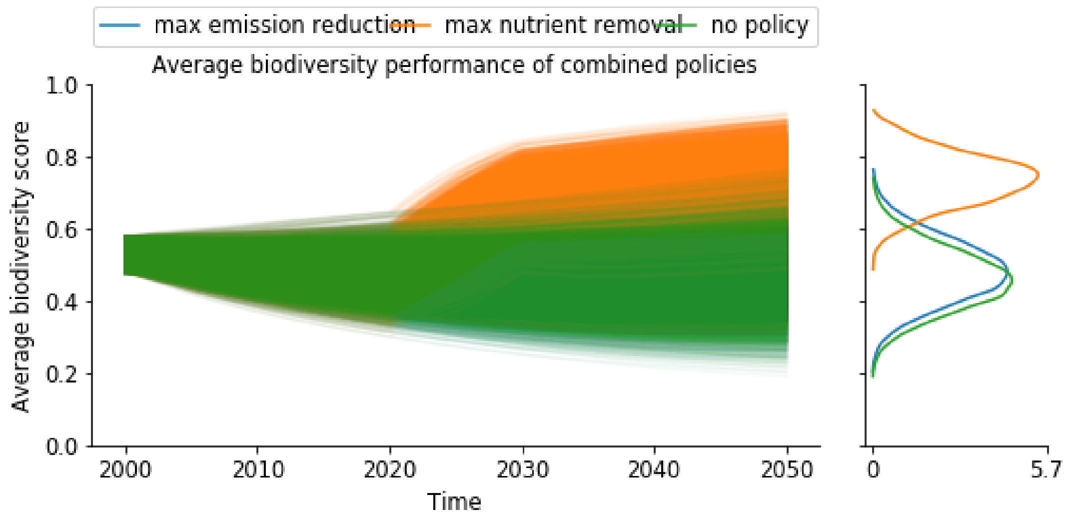

Figure 13: Average biodiversity scores for combined policy measures.

When policies for emission reduction (e.g. *Livestock*, *Manure* and *Agriculture transition*) and nutrient removal (e.g. *Landscaping*, *Turfing* and *Dredging*) are combined a clear distinction in biodiversity outcomes can be observed (Figure 13). Here nutrient removal policies significantly outperform emission reduction policies in terms of biodiversity improvement. When nutrient emission reduction policies are combined, however, a slight improvement of average biodiversity is noticeable. The same behaviour is observed for terrestrial biodiversity scores, as presented in Appendix B. However, the performance of *max nutrient removal* improvement of aquatic biodiversity remains limited (See Appendix B).

Cost comparison of policies

The cost performance of the individual policy measures are displayed in Figure 14. The figure shows that the total policy costs of livestock reduction and manure application are the outliers, adding up to an estimated 13 and 22 billion, respectively. The other policy measures are limited to a total cost of 10 billion, with agriculture transition measures scoring the lowest at a total of around 5 billion.

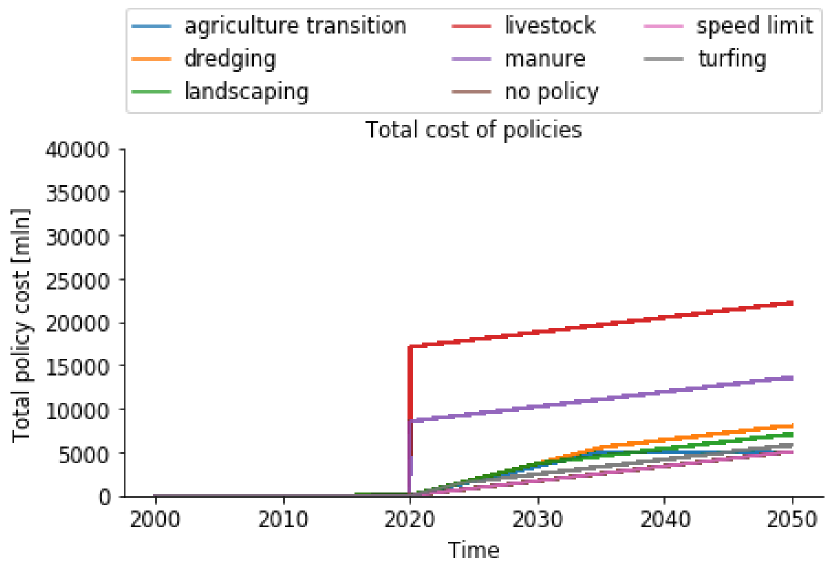

Figure 14: Cost comparison for individual policy measures to improve biodiversity.

The combined policies for emission reduction (e.g. *Livestock*, *Manure* and *Agriculture transition*) or nutrient removal (e.g. *Landscaping, Turfing* and *Dredging*) show vastly different policy cost performance (Figure 15). The combined emission reduction policies almost triple the total policy cost required for nutrient removal until the year 2050. Note that the cost for emission reduction of manure and livestock are based on a "worst case" cost scenario, which estimates a total cost of 17 billion for a 50% reduction of livestock (Kuiper & Rutten, 2021). In a "best case" scenario, the total cost would be around half of the worst-case, adding up to around 15 billion. Note that in this case the combined nutrient removal policy would still outperform the combined policies for emission reduction on total policy costs.

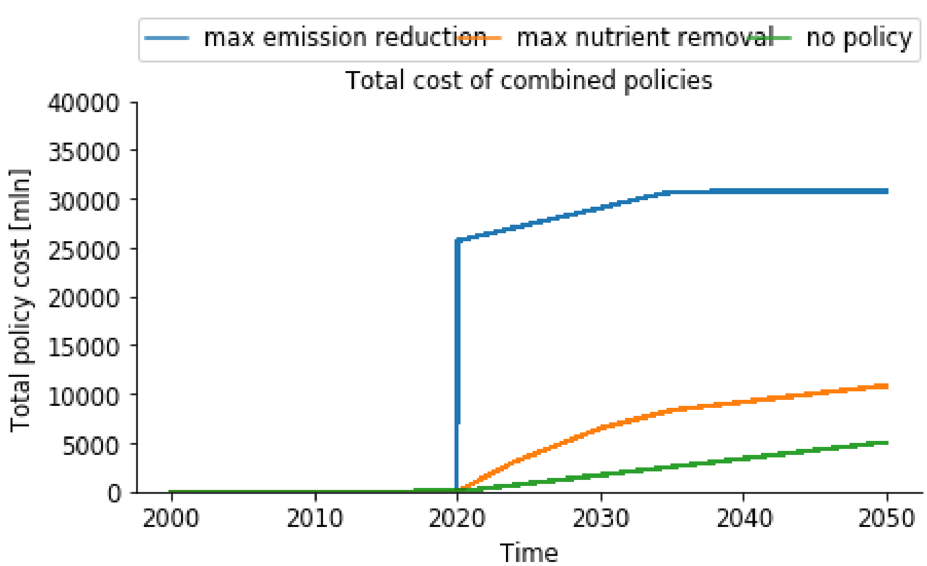

Figure 15: Cost comparison for combined policy measures to improve biodiversity.

4.3.2 EU guidelines policy approach

EU guidelines are setup for the preservation of air quality standards (European Union, 2016). To this end, the EU guidelines provide limits for domestic emissions of NH_3 and NO_x for the year 2030. The reduction goals are determined for the year of 2030 with respect to emissions in 2005. When testing the ability of policies to meet the EU guidelines, it stands out that none of the policies outperform the *no policy* option (Figure 16). From figure 14 can be deduced that the *no policy* option has the most low policy cost outcomes. Only the policy for manure reduction outperforms the no policy option in terms of avoiding worst-case cost scenario's.

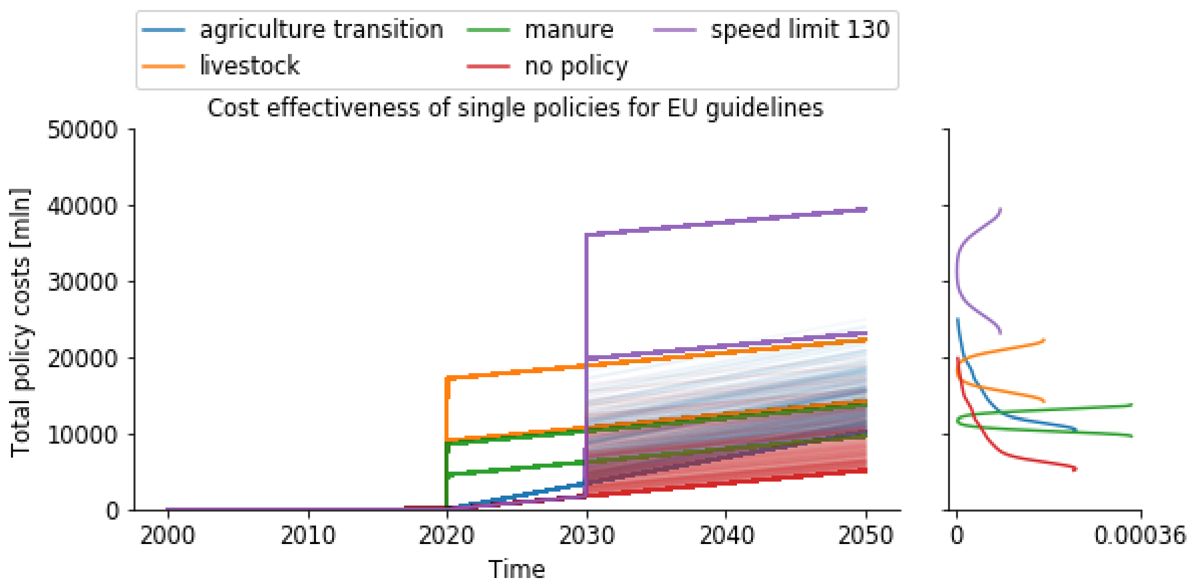

Figure 16: Total policy costs of individual policies to meet EU guidelines.

Especially noteworthy is the bad performance of a higher speed limit. The impact of a higher speed limit is representative for higher NO_x emissions. This is in line with findings in paragraph 4.2.2, which shows that the largest issue for compliance with EU guidelines is domestic NO_x emissions from traffic. The results also show that a lower speed limit of 80, would ensure that the EU guidelines are met across the entire space of uncertainties (See appendix C). Lowering the speed limit further down to 80 km/h is thereby the most reliable policy option to meet the EU guidelines.

The policy options that aim to reduce domestic emission related to agriculture (e.g. *Livestock* and *Manure*) are reliably able to meet the EU guidelines (Figure 17). The majority of cost outcomes for these policies are higher than the *no policy* option. The less drastic implementation of livestock policy, which reduces livestock with only 25% instead of 50%, shows to be reliably able to meet EU guidelines (Figure 17). Here, a 25% livestock reduction shows similar performance as the 50% manure reduction policy. A less drastic implementation of the manure policy of 25% instead of 50%, however, indicates to lose its reliability in meeting EU guidelines (Figure 17). When manure is only reduced by 25% a variety of scenarios still exist which require significant additional emission reduction.

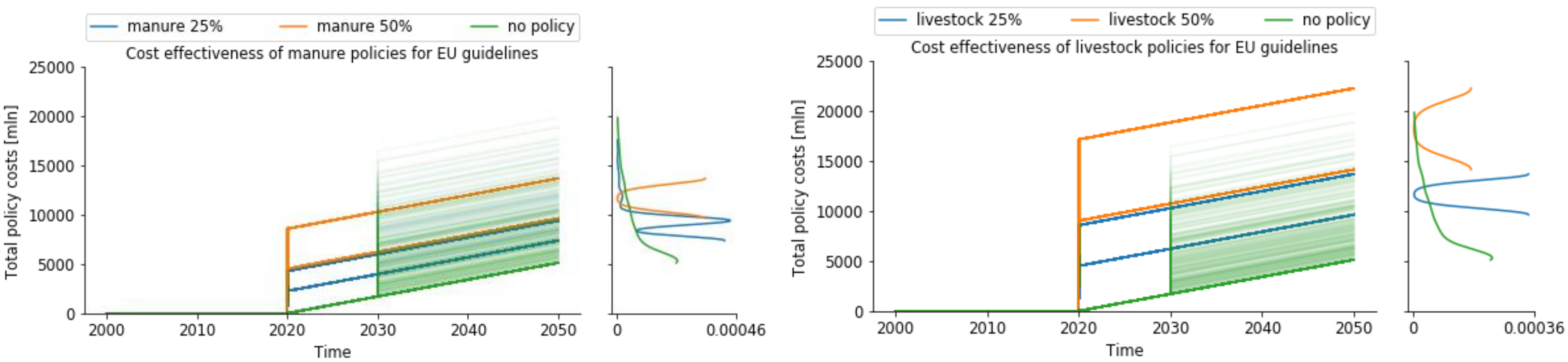

Figure 17: Cost comparison of different levels of manure and livestock policies to meet EU guidelines.

Based on these findings, the policy costs for manure and livestock reduction are deemed disproportional costly relative to the marginal improvement of reliability. As also shown in Appendix C, NO_x reductions from manure and livestock reduction are not sufficient to bring the speed limit back up to 130 km/h within the context of EU guidelines. The recommendation for meeting EU guidelines is to aim policy at reducing the uncertainty related to high domestic NO_x emissions from traffic (Paragraph 4.2.2). This could be accomplished by ensuring an increased outflow of gas and diesel cars from the fleet and ensuring successful transition policy from government through subsidies. In case such interventions are found to be unsuccessful, a further reduction of the speed limit is a reliable option to meet EU guidelines (See appendix C).

4.3.3 NL guidelines policy approach

NL guidelines for emission reduction are setup for the preservation of Natura 2000 areas (Adviescollege Stikstofproblematiek, 2020). The NL guidelines are not setup specifically for NO_x and NH_3 emissions. Instead, the ambition for emission reduction is determined in terms of total nitrogen reduction with respect to 2019 (Adviescollege Stikstofproblematiek, 2020). The ambition is to reduce Nr emission by 30% in 2030 and 50% in 2035. Figure 16 shows the effectiveness of policies to meet NL guidelines in terms of costs. From Figure 16 can be observed that livestock reduction, manure reduction and a speed limit reduction to 80 km/h, are able to consistently outperform the *no policy* option. However, only the livestock reduction policy is able to reliable avoid worst case cost scenarios. Noteworthy is that manure reduction is able to ensure low policy cost outcomes in a majority of scenarios. However, manure reduction still results in a significant amount of high cost scenarios, as can be deduced from the tail in the density function (Figure 18). Lastly, a lowering of the speed limit to 80 km/h shows a reliable but marginal improvement over the *no policy* option.

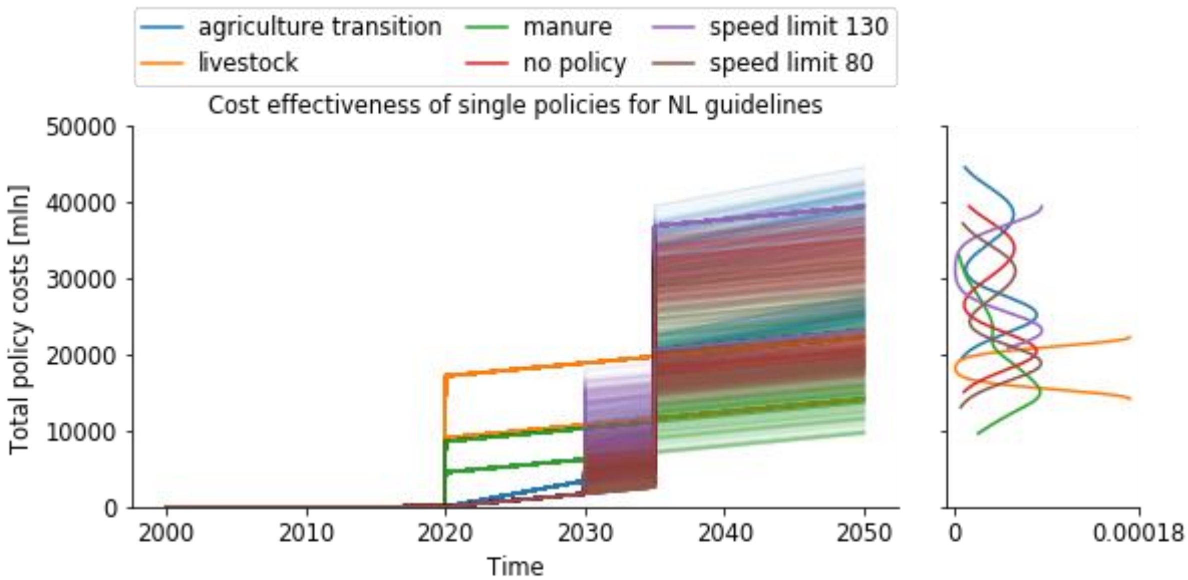

Figure 18: Cost comparison of individual policies to meet NL emission guidelines.

Agriculture transition and speed limit show to be ineffective at meeting NL guidelines. Obviously, an increase of the speed limit to 130 km/h is a deterioration of circumstances, compared to the current case with a speed limit of 100 km/h. Figure 18 does show that the impact of an increase of speed limit results in a drastic increase of worst case cost scenarios. The finding shows a vulnerability of the NL guidelines to an increase in speed limit to 130 km/h, and that the reduction of the base policy to 100km/h is necessary.

The policy for agriculture transition also shows to be ineffective at meeting NL guidelines, and results in worse results than the *no policy* option (Figure 18). This can be explained by the lower potential for emission reduction from agriculture that results from additional agriculture transition policy. In the cases where NL guidelines are not met, the lower potential for emission reduction results in a larger requirement for livestock reduction. Consequently, the policy cost to meet NL guidelines increase after the implementation of agriculture transition policy compared to the *no policy* option. This reverse effect of agriculture transition on the policy costs outcomes is thus due to a modelling choice. This modelling choice is therefore biased against the agriculture transition policy option, making it a limitation of the model.

When considering the most successful policies for meeting NL guidelines in more detail (e.g. *Livestock* and *Manure*), it can be observed that only a 50% reduction of livestock is reliably able to meet the NL guidelines (Figure 19). A 25% reduction of livestock shows similar cost outcomes as a 50% manure reduction, where both still result in a significant amount of high cost outcomes. A 25% manure reduction shows a large deterioration of performance compared to a 50% manure reduction with a large increase in high cost scenarios. Indicating that milder manure or livestock policies result in significant loss of reliability.

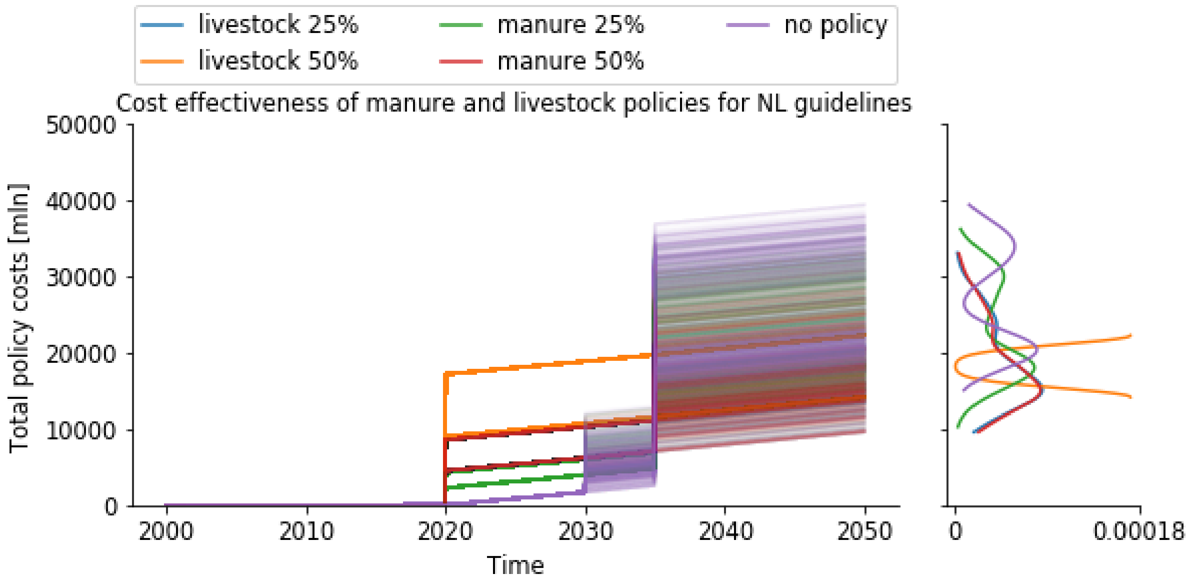

Figure 19: Cost comparison of manure and livestock policies to meet NL emission guidelines.

4.3.4 Adaptive policy approach for NL guidelines

Within the adaptive framework, a feedback between the performance of terrestrial biodiversity and NL emission guidelines is incorporated. The emission guidelines are linked to the state of terrestrial biodiversity because most KDW sensitive areas are terrestrial (See Appendix P). In case biodiversity in terrestrial Natura 2000 areas improves, the NL emission guidelines are partially relaxed. On the contrary, in case biodiversity in terrestrial Natura 2000 areas declines, the NL emission guidelines become stricter. Due to the incorporation of biodiversity performance, the successful ecological policies from paragraph 4.3.1, (e.g. *Landscaping* and *Turfing*) are also implemented in the adaptive policy approach.

From Figure 20, it is observable that only the policy options of *Landscaping*, *Livestock* and *Manure* result in consistent improvements over the *no policy* option. *Turfing* shows to be ineffective at improving past the *no policy* norm. This can be explained by the fact that *Turfing* is ineffective at improving biodiversity on the short term. Similar to findings in 4.3.3, *Livestock* is the only policy option which reliably avoids high cost scenarios. *Manure* and *Landscaping* also result in low cost outcomes in the majority of scenarios (Figure 20). The downside of these policy options is that there still is a significant risk of high cost outcomes in a variety of scenarios. Manure reduction and landscaping policies thereby lack the individual robustness to avoid high cost outcomes, that livestock reduction does possess.

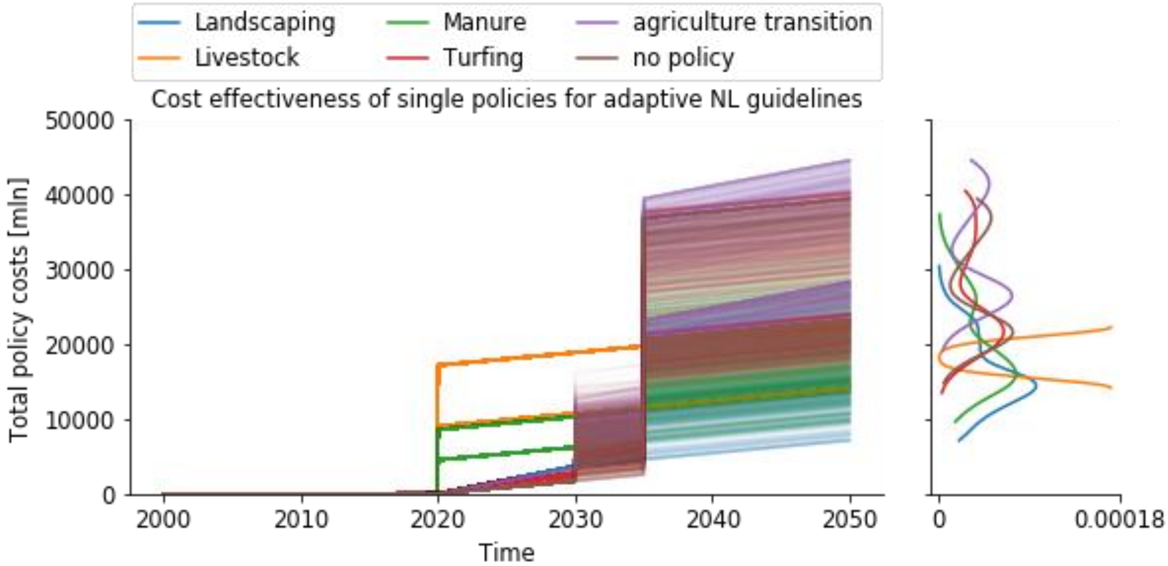

Figure 20: Cost comparison of individual policies to meet adaptive NL emission guidelines.

When combining the effective policies for emission reduction for adaptive NL guidelines (e.g. *Livestock* and *Manure*) with *Landscaping*, a strong improvement of cost outcomes is observable (Figure 21). Figure 21 shows that a combination of landscaping and manure reduction results in reliable avoidance of high cost outcomes. The 50% livestock reduction policy was already shown to be reliable for the avoidance of high cost scenarios for non-adaptive NL guidelines (Paragraph 4.3.3). The addition of landscaping to the 50% livestock policy is thereby not of added value. However, when considering a lower level of livestock reduction, equal to 25% instead of 50%, the combination with landscaping does pay-off (Figure 22).

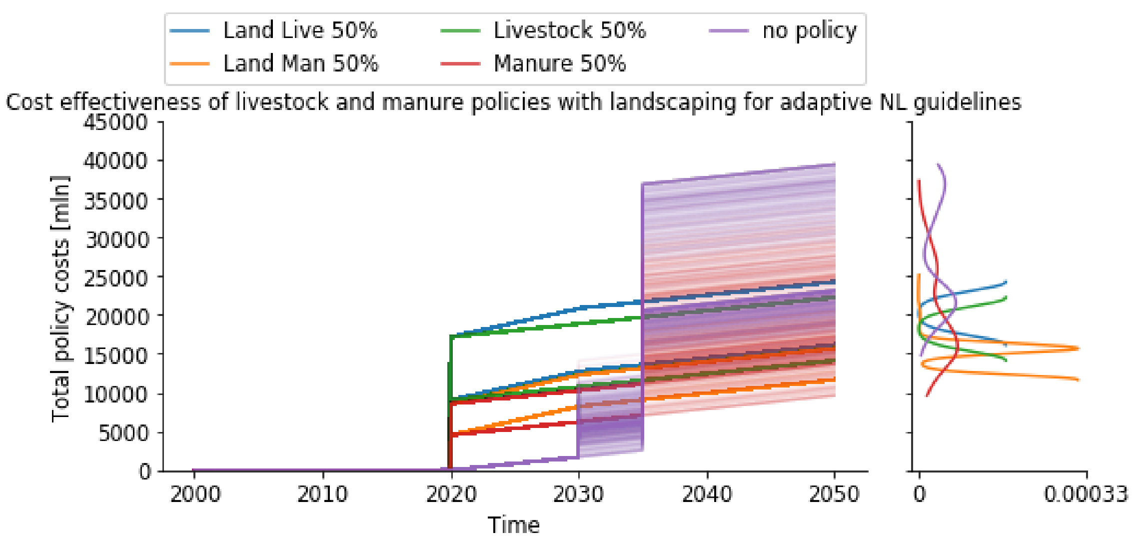

Figure 21: Comparison of cost effectiveness of livestock and manure policies in combination with landscaping.

Figure 22 indicates that a combination of 25% livestock reduction and landscaping, results in reliably lower cost outcomes than a 50% reduction of livestock.

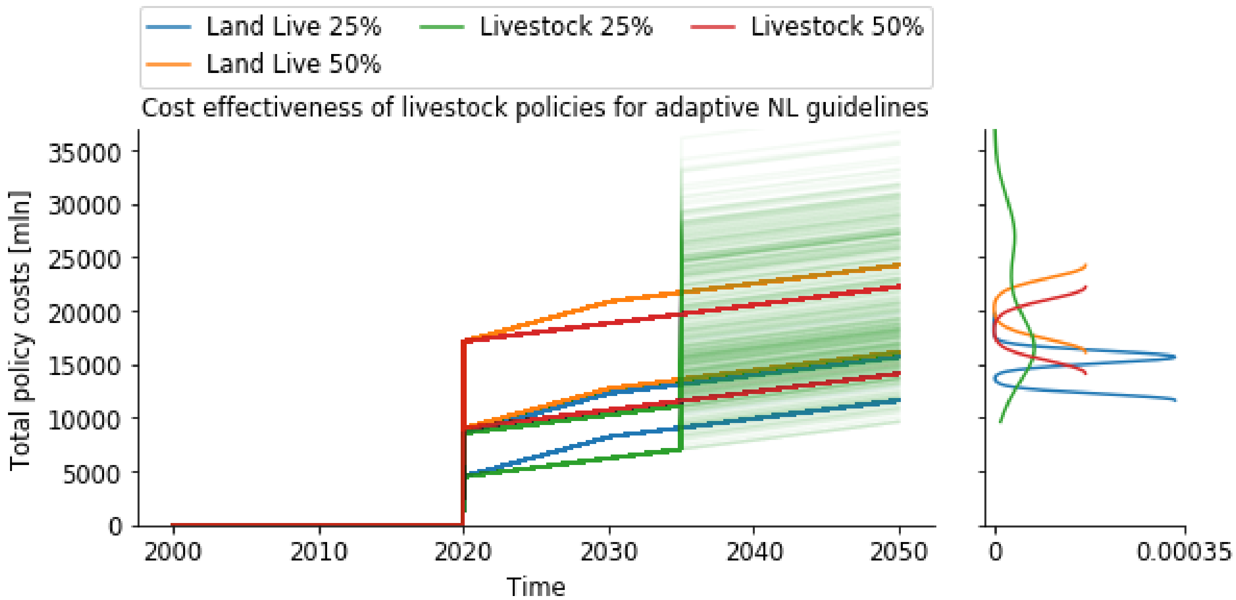

Figure 22: Policy cost comparison of different levels of livestock reduction combined with landscaping.

The combination of landscaping and manure reduction also shows significant improvement in cost outcomes (Figure 23). The combination of 50% manure reduction with landscaping results in fairly reliable avoidance of high cost outcomes. The combination of a 50% manure reduction and landscaping does still result in some scenarios that exceed the 20 billion mark. The combination of landscaping with a 25% manure reduction shows a majority of cost outcomes around the 10 billion mark. The combination does still show a sensitivity for certain scenarios that result in high cost outcomes, as indicated by the tail that can exceed the 25 billion mark (Figure 23). Manure policy therefore lacks robustness to avoid high cost outcomes, even in combination with landscaping policy.

When comparing the performance of the combined policies of manure and livestock reduction with landscaping, the combination with livestock shows superior robustness (Figure 22 & 23). As presented in Figure 22, the combination of livestock reduction with landscaping shows no sensitivity for higher cost scenarios. Moreover, the combination of landscaping and a 25% livestock reduction is reliable enough to allow for the speed limit to be increased back to 130 km/h, without violating the adaptive NL guidelines (See Appendix E). It is therefore recommended for explore opportunities to make adaptive NL guidelines a reality. If successful, landscaping measures can be used to improve biodiversity and can result in robust and favourable cost outcomes to the nitrogen crisis.

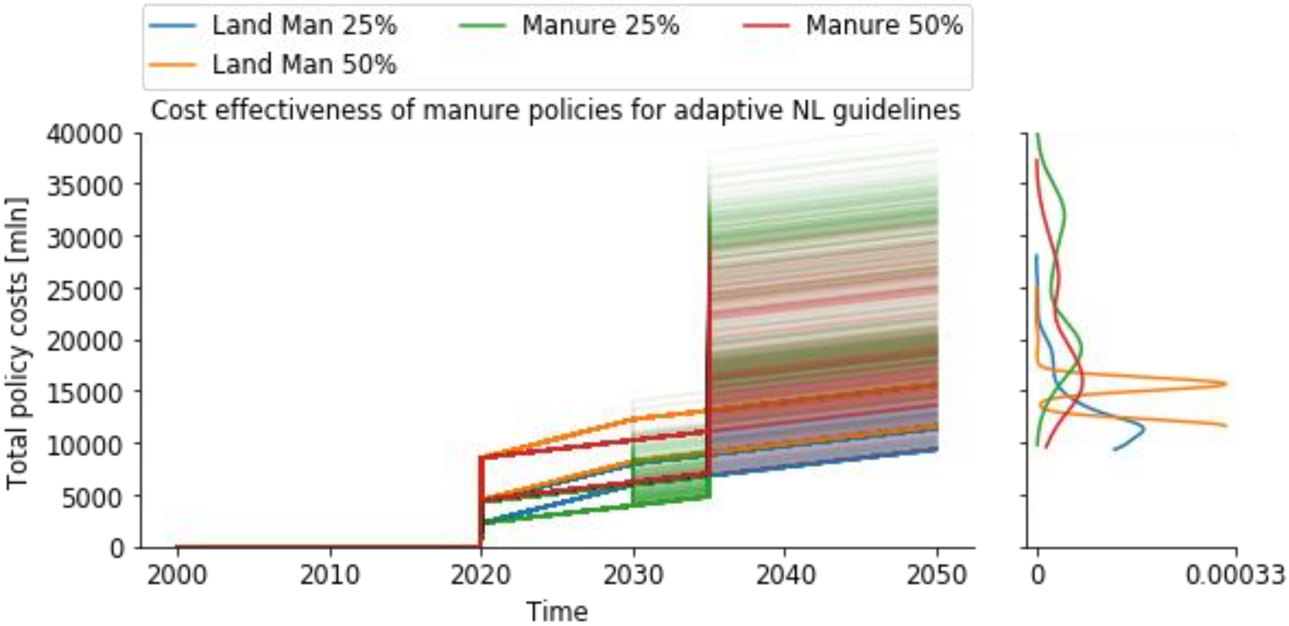

Figure 23: Policy cost comparison of different levels of manure reduction combined with landscaping.

5. Political reflection

The results show that ecological policies outperform policies that are aimed at emission reduction in terms of improving biodiversity (Paragraph 4.3.1). Also, ecological policies show potential for improving the cost outcomes related to NL guidelines (Paragraph 4.3.4). Based on the findings of this research, the current policy focus on emission reduction is deemed ineffective from an ecological perspective, resulting in inefficient policies from a cost perspective. The results of this research therefore indicate that a shift in the policy approach is required to ensure economically efficient and ecologically effective outcomes.

An adjustment in the policy approach based on the results of this research requires two main barriers to be overcome. First, a shift in policy approach requires the results of the model to be accepted in the political arena, before further actions can be taken. Second, a feedback structure between the state Natura 2000 areas and the Dutch nitrogen emission guidelines must be implemented in the regulatory framework. This is a base requirement, because if ecological measures cannot relax emission guidelines, an ecological approach cannot provide solutions to the nitrogen crisis.

5.1 Modelling approach in the political context

For the findings of a modelling approach to incite a change in policy approach, it must be accepted into the decision-making process. To this end, the two other major forces in the political context have to be considered, namely: empiricism and policy (Figure 24). What a model can do is explore new solution avenues and provide a proof of principles for these avenues. But without empirical substantiation, model results do not hold up in political discourse. Only when empirical evidence is found, can the model's claims be substantiated and implemented in the regulatory framework.

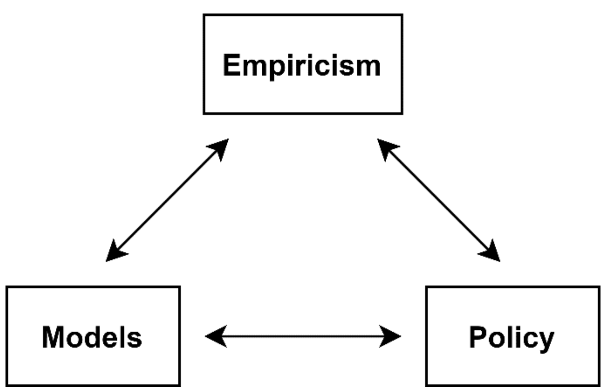

Figure 24: Models in het political context

With the Nitrogen Crisis currently at a gridlock, it gives way to societal tension and uncertainty in the business sector (Stokstad, 2019a; Vrieselaar & Barendregt, 2021). Policy options for livestock reduction seem necessary to resolve the crisis (Kuiper & Rutten, 2021), but are undesirable due to its impacts on the livelihood of farmers and resulting in massive loss of economic activity . The modelling approach of this research indicates that an ecological policy approach to the Nitrogen Crisis potentially has enormous societal benefits. This research thus provides the bases from which empirical scientists and policy makers can further investigate an alternative approach to the Nitrogen Crisis

Policy makers and scientists can use the findings of this research to start exploring the feasibility of an ecological approach to the Nitrogen Crisis. Based on this research, policy makers can start adding points to the agenda to stimulate the development of the field of ecological engineering. Moreover, policy makers can start exploring judicial pathways to get ecological measures accepted into the regulatory framework. Additionally, policy makers can start setting up the necessary regulatory base for the implementation of ecological measures. Scientists can use the proof of principle from this research to start exploring the feasibility of ecological measures. Moreover, they can use this research to find financial and academic stimulus for their research.

5.2 Barriers and opportunities of the regulatory and legislative context

To realize a shift towards an ecological policy approach, the approach must be integrated into the existing legislative framework. Within the context of the Nitrogen Crisis, strict emission regulations and laws concerning nature preservation must be considered. For starters, EU emission guidelines are crucial as they define emission ceilings for domestic emissions (European Union, 2016). These EU emission guidelines cannot be avoided through ecological improvement as they are setup to maintain air quality standards, which is a public health issue. The EU guidelines are thus rigid and cannot be circumvented or adapted based on ecological measures.

Contrary to EU emission guidelines, Dutch nitrogen emission guidelines are directly related to the state of nitrogen sensitive Natura 2000 areas (Aanpak Stikstof, 2021; Adviescollege Stikstofproblematiek, 2020). The aim of the Dutch nitrogen law is to ensure that 50% of the nitrogen sensitive area is brought below the KDW by 2030, and 74% by 2035. Based on these goals, nitrogen emission reductions of respectively 30% and 50% compared to 2019 are setup. Ecological measures could be used to improve the state of nitrogen sensitive areas. If successful this could result in lower emission restrictions, and the avoidance of unnecessary high policy costs related to emission reduction.

Implementing ecological measures with the aim of reducing emission guidelines must consider the Dutch Nature law, as established in 2009 (Overheid, 2015). The Dutch nature law is based on the European bird and habitat directive (European Commission, 2000). To determine the feasibility of an ecological policy approach, article 6, sections 1 to 4 of the habitat directive must be considered . The major obstacle which cannot be circumvented by policy makers is section 3. Article 6 section 3 indicates that any project that is likely to have an impact on a Natura 2000 site has to undergo an Appropriate Assessment (European Commission, 2000). Only if the assessment ascertains that the project will not have any adverse effects on the integrity of the site, may the project continue. Meaning that no new project that results in nitrogen depositions can continue unless these areas have been brought below the KDW. Alternatives for the KDW approach lack scientific backing, meaning that KDW's are likely going to stay the norm for Nature preservation policy for the years to come (Kamerbrief, 2021). Opportunities for the relaxation of emission guidelines are however present in paragraph 1 and 4 (European Commission, 2000).

Article 6(1) indicates an obligation to bring Natura 2000 areas into a favourable conservation status. If ecological policies can do so, the nitrogen resilience of these areas could be increased, making the areas less vulnerable to nitrogen pollution. This could allow for emission guidelines to be relaxed and thus more projects to continue. It is however unlikely that areas can be given an new KDW values, since KDW's are assigned based on habitat type (H.F. van Dobben & A. van Hinsberg, 2008). Article 6(1) could potentially hold up if the state of the area has been improved to the point where an increase of nitrogen pollution does not have a degenerative effect on the area. In this case, an Appropriate Assessment, as discussed in Article 6(3) needs to be made to prove that the area has an increased nitrogen resilience and that additional nitrogen depositions are not harmful to the area.

Another promising route for the use of ecological measures is the one described in article 6(4) (European Commission, 2000). This article describes that under exceptional circumstances projects with a negative assessment may continue. These circumstances relate to issues of overriding public interest, including reasons of social and economic nature, where there are no readily available alternatives. In such cases the Member State must take appropriate compensatory measures to ensure that the overall coherence of the Natura 2000 Network is protected (Article 6.4). Article 6(4) thereby refers to the possibility of compensatory measures. Such compensatory measures can refer to ecological measures. The use of which would require scientific substantiation and evidence for overriding public interest for it to be legally accepted. In the case of the nitrogen crisis, ecological measures could be used to compensate for livestock pollution. And the argument of overriding public interest could be made based on the economic and societal importance of its preservation.

5.3 Importance of scientific substantiation of ecological measures

Dutch emission guidelines can only be relaxed based on Article 6(1) or 6(4) if scientific substantiation of ecological measures can be provided. Currently the legislative framework is not ready for the new policy approach due to a lack of scientific substantiation. Empirical evidence is lacking on the ability of ecological engineering measures to compensate any potential negative impacts of nitrogen emission projects. Empirical research on the feasibility of ecological engineering as a compensatory measure is thus required for the realization of an ecological policy approach.

Focussing efforts on legitimizing ecological measures could thereby break open the discourse from a focus on emission reduction towards a focus on ecological improvement. This would be especially useful as currently the focus of emission policies is heavily focussed on the agriculture and construction sector. Especially in the agriculture sector policies for expropriation and buy-outs have resulted in strong societal backlash (Stokstad, 2019a). Broadening the solution space could thereby help relieve the pressure on this sector and avoid societal unrest. Moreover, the potential benefits for an ecological engineering approach can result in significant economic benefits. Large parts of the agriculture sector could be kept operational, maintaining large parts of its yearly economic contribution and avoiding costs for buy-outs and expropriations.

The societal and economic benefits of an ecological approach are potentially enormous and provide a strong incentive to develop the scientific field of ecological engineering. Currently, the findings in the field of ecological engineering to offset emissions and improve ecology are no to promising. Investment in this field thus has the risk of not acquiring the scientific backing that is required to allow for relaxation of emission guidelines through ecological compensation. The potential economic benefits could however incentivize significant investments from government, potentially creating a breakthrough in the field. If so, scientific substantiation must be realized before the emission reduction policies are enforced by the Dutch government. Meaning, ecological compensatory measures should be legally accepted well before the year 2030 and 2035 to compensate for the respective emission reduction goals.

6. Discussion & Conclusion

The research was conducted with the aim to investigate the benefits of a wider ecological framework for the mitigation of the nitrogen crisis in the Netherlands. In this chapter the results are discussed within the context of the relevant literature and regulatory framework. Additionally, the utility of the model results for the political discourse is discussed. Moreover, the limitations of the model are discussed, followed by recommendations for further research.

6.1 Research results

This research has integrated the stores, fluxes and cycles of N and P, through both terrestrial and aquatic ecosystems, and has incorporated a link with the state of biodiversity. According to Guignard (2017), the integration of stores fluxes and cycles is critical for a better understanding of the macronutrient fate of N and P nutrients. The understanding is further aided through the integration of both terrestrial and aquatic ecosystems and the interaction between the nutrient cycles (Grimm et al., 2003; Guenet et al., 2010; Soininen et al., 2015). The interaction of N and P nutrients are foundational for the understanding processes of underlying plant growth (Chapin III et al., 2011; Elser et al., 2007; P. M. Vitousek et al., 2010a), and consequently to the state of biodiversity and ecosystems (Elser et al., 2009, 2010). The integration of N and P in this research is deemed essential to understand the macronutrient fate of N and P, and ultimately to maintain agriculture productivity and the provision of ecosystem services (Guignard et al., 2017).

Through the integration of the N and P cycles and its link to biodiversity, this research has extended the policy framework for the mitigation of the Dutch nitrogen crisis. The extended framework allows for a broader analysis of the systems vulnerabilities and policy interventions. The results show that the performance of biodiversity is highly dependent on the impact of nutrient availability (Paragraph 4.2.1). Ecological policy measures are shown to reliably result in higher biodiversity outcomes compared to emission mitigation measure (Paragraph 4.3.1). In addition, the implementation of ecological measures resulted in drastically lower policy costs than those related to emission reduction policies (Paragraph 4.3.1). The insights of this research indicate a potential for emission mitigation through ecological compensation, that can result in the preservation of large parts of the multi-billion agriculture industry.

These results indicate that highest potential for biodiversity improvement is related to the dynamics of nutrient stocks within ecological systems. The current policy focus of the Dutch government on emission reduction (Adviescollege Stikstofproblematiek, 2020) is therefore regarded as a form of symptom control that diverts attention from the root of the problem, namely; ecological degradation. The focus on emission reduction in the decision-making process is also manifested by the use of emission and deposition models (Adviescollege Stikstofproblematiek, 2020), such as the OPS (Sauter et al., 2020) and EMEP model (Pisoni et al., 2019). The extended framework of this research shows how these models lack the broad integration of nutrient cycles and ecology that is required to effectively design policies for nature preservation.

6.2 Barriers and opportunities for the use ecological measures

For the implementation of a new ecological policy approach a shift in political discourse is required, To accomplish a shift in political discourse, the results of this research must be put into action in the political arena and needs to be accepted into the decision-making process. To effectively steer the political discourse, the modelling approach is dependent on policy makers and empirical scientists. For the model to be effective, it thus has to incite policy makers and empirical scientists to action. This can be achieved by conveying the gigantic problem solving potential and societal benefits of an ecological approach to the nitrogen crisis, that is indicated by the modelling approach. If successful, it will incite policy makers to change the agenda towards ecological exploration and incite scientists to conduct empirical research. When both are accomplished and produce favourable results, an ecological problem solving approach to the nitrogen crisis can become feasible.

A shift in policy perspective, away from emission reduction and towards ecological measures, is faced with scientific and legal barriers. These barriers are related to a lack of scientific substantiation of ecological measures and the legally established guidelines for nitrogen emission reduction. If scientific substantiation of ecological compensatory measures is present, these measures can be used as compensation for new projects that result in nitrogen deposition in nitrogen sensitive Natura 2000 areas (European Commission, 2000). The inability of new project to continue is a central issue of the Dutch nitrogen crisis, making the lack of scientific substantiation an important barrier. Moreover, the legally established nitrogen emission reduction guidelines by the NL and EU cannot be disregarded when considering a shift in policy perspective. These guidelines are legally binding and are likely to require significant reduction of existing nitrogen emission sources with huge economic impact.

For ecological measures to provide a solution to the nitrogen crisis, they need to be legally accepted as a compensatory measure for N emitting projects, or must have a relieving effect on emission guidelines. The EU guidelines do not provide possibilities for relaxation through ecological measures, as these guidelines are aimed at preserving air quality standards. The EU guidelines can thus not be omitted through improved ecological performance (European Union, 2016). Fortunately, the results of this research showed that the EU guidelines are not the bottle neck for the nitrogen emission issue (Paragraph 4.3.2). The real bottle neck for the nitrogen crisis results from the NL emission guidelines, which are likely to require significant reductions of the agriculture sector for compliance (Paragraph 4.3.3). The NL emission guidelines provide opportunities for guideline relaxation by means of ecological compensation. This is the case as NL emission guidelines are determined based on KDW's, which are a measure of the nitrogen resilience of Natura 2000 areas (Adviescollege Stikstofproblematiek, 2020).If ecological measures are thus able to significantly increase the resilience of Natura 2000 areas to the points where additional nitrogen depositions are not harmful, new project can resume based on Article 6 section 3 of the habitat directive.

There are indications that ecological measures are effective at increasing ecological resilience to nitrogen pollution (Provincie Gelderland, 2017). However, empirical evidence is lacking to support claims that ecological measures can increase the nitrogen resilience of Natura 2000 areas to a point where KDW exceedance no longer results in ecological degradation. If such scientific backing can be realized, ecological measure could be used to compensate for nitrogen deposition. When realized, it would allow for the conservation of large parts of the multi-billion agriculture sector and avoid societal unrest related to expropriation and buy-outs. The potential societal benefits of an ecological approach to the Dutch nitrogen crisis are therefore deemed tremendous.

6.3 Research recommendation

The System Dynamics model of this research is setup to; model the flow of nutrients through air, soil and water, analyse the impact of N and P nutrients on biodiversity and determine the impact of a variety of policies and emission guidelines. The wide array of subjects that the model covers makes it prone to limitations. The most relevant model limitations related to the modelling of biodiversity, the flow of nutrients and economic impact are discussed in this section. Additionally, this section discusses the avenues for further research outside the scope of modelling approach. These research avenues are aimed at empirical and judicial studies that are required for the practical implementation for an ecological policy approach to the Dutch nitrogen crisis.

6.3.1 Improvements of the modelling approach

As discussed in the validation (Paragraph 3.5) the main areas for improvements of the modelling approach are related to the ecological complexity, the flow of nutrients and the economic dimension. In this section, these areas of improvement of the modelling approach are discussed in further detail.

Ecological complexity

A main limitation of the model is related to the ecological complexity underlying the development of biodiversity. First, in the model the measure for biodiversity is determined based on the ratio between desirable and undesirable plants. This approach is likely overly simplistic to represent the complexity of biota that exists in the different types of Natura 2000 habitats. Moreover, the state of biodiversity is subject to strong assumption on the initial state of biodiversity and the utilization of biomass potential in Natura 2000 areas. Assumptions also had to be made relating to the distribution of in-land aquatic Natura 2000 areas over the identified waterbodies in the model. With strong differences in the state of biodiversity in each of the waterbodies, this assumption has a strong influence on the aquatic measure of biodiversity.

The model's limitation of representing ecological complexity is also related to the nutrient dynamics underlying biomass growth. This dynamic is based on the differentiation between the biomass growth of either desirable or undesirable plants. The growth of either biomass is based on two forms of nutrient dynamics, namely: nutrient limitation and nutrient availability. The implementation of the nutrient dynamics in the model is subject to the assumption that terrestrial ecosystems are nitrogen limited (P. Vitousek & Field, 2001), whilst aquatic ecosystems are phosphor limited (Djodjic et al., 2004). These assumptions are in line with literature, but neglect cases of phosphor limited terrestrial ecosystems (P. M. Vitousek et al., 2010b) or nitrogen limited aquatic ecosystems (Rabalais, 2002). Moreover, the modelling approach neglects the possibility of a shift in limitation due to nitrogen and phosphor interaction (Ågren et al., 2012; Elser et al., 2010). Moreover, the impact of the nutrient availability on biomass growth is modelled based on impressionistic impact graphs, which are constructed based on a limited amount of scientific evidence. Nutrient availability was also shown to be highly predictive of low biodiversity outcomes (Paragraph 4.2.1). These findings indicate that this key element of the model is not yet well understood, and should be researched further for a better understanding of the role of nutrient availability for the development of biodiversity.

The lack of ecological complexity in this model also influenced the level of detail that ecological policies could be tested on. Especially the ecological measure *Landscaping* was implemented simplistically and is unable to represent the entire space of possible ecological measures. The simplistic ecological approach has likely overestimated the effectiveness of the landscaping to increase biodiversity. Still, the results indicate the ecological policy measures show great potential for ecological improvement when compared to emission reduction policies. But before the research can be conclusive, the ecological policies must be worked out in more detail to align with the ecological complexity underlying biodiversity.

Flow of nutrients

The flow of nutrients throughout the different mediums is subject to large degrees of complexity which had to be significantly reduced for this research. The transportation of nutrients through the soil for instance does not incorporate the impact of different soil types (e.g. clay, sand, loam etc.), which are shown to be significantly influential for the degree of leaching that occurs (Djodjic et al., 2004). The simplification of soil types also neglects potential differences in soil concentrations (Djodjic et al., 2004) and its impact on different types of biota (Barrios, 2007). Similar simplification had to be made for the modelling of nutrient flows throughout the Dutch river delta due to a mismatch in data. The assumptions relate to the distribution of water flows over the determined waterbodies and the nutrient concentration of the specific waterbodies. Also, the dynamic relation between soil and water is simplified into a one-way street where nutrients only flow from soils to water. The model thus does not incorporate possible sedimentation or diffuse processes which allow nutrient to be bilaterally exchanged between water and soils (Klump & Martens, 1981). These assumptions influence the flow and accumulation of nutrients which are foundational to the state of aquatic biodiversity.

The transportation of nutrients through the soil, air and water are impacted by meteorological variability (Cuhadaroglu & Demirci, 1997; Sharpley, 1997; Zhou et al., 2016). The variability of rainfall is not incorporated in the model, which assumes a constant flow of water throughout the Dutch river delta. Moreover, the impact of wind and temperature are excluded in the model. The processes underlying nutrient leaching and run-off, and arial processes of acid rain and phosphoric dust are over simplified due to the exclusion of meteorological factors. These factors are potentially influential for the development of biodiversity and should thus be further implemented to gain a better understanding of their influence on biodiversity development.

The flow of nutrients throughout the atmosphere is based on the assumption that deposition of arial nitrogen occurs uniformly distributed over the Dutch surface area. This assumption neglects the geographical nature of emissions and depositions, which are vastly different NO_x, NH_3 and phosphate dust

(Adviescollege Stikstofproblematiek, 2020). The model is thus unable to distinguish critical point source of emissions which could be highly problematic for the compliance to region specific KDW values. Moreover, the exclusion of geographical emissions and depositions disallows the detailed investigation of the uncertainties related to the foreign inflow of nitrogen. Such inflows provide a major uncertainty to KDW compliance, but cannot be accurately represented without the geographical integration of emission and depositions. The exclusion of a geographical dimension is thus found to be an important limitation of the model and should be incorporated further for a more effective design of emission mitigation policies.

Economic dimension

The economic dimension of the research is implemented from a simple investment cost perspective. The cost of policies are thereby determined based on the expected investment cost for its implementation. In the regulatory policy approach the policy investments costs also incorporates the additional cost for livestock reduction. Here, the required amount of livestock reduction is determined based on the additionally required emission reduction to meet EU or NL guidelines. This cost approach has the fundamental issue of only considering direct investments costs. It thereby neglects the indirect costs related to reduced economic activity, through intermediate economics and the multiplier effect (Beattie, 2021; Hindriks & Myles, 2013). This is especially problematic in the case of the reduction of agriculture practices. On a yearly bases the agriculture sector is responsible for over 30 billion in added value (Adviescollege Stikstofproblematiek, 2020). The loss of economic activity is therefore substantial but is not represented by the total policy costs. Further research should therefore make a more comprehensive economic analysis of policies.

Moreover, the reduction of the livestock sector has impact on the production of manure, which is a valuable resource (Leenstra et al., 2019). The reduction of livestock reduces the production of manure, which is vital for the cultivation of crops (CBS, 2020b). The additional costs for manure import due to livestock reduction are thereby neglected. A reduction of livestock also has an impact the demand for agriculture crops, since a large part of domestic plant produce is used as rough feed for livestock (CBS, 2020b). These economic implications related to the reduction of agriculture practices are neglected. Instead cost estimations are based on reports for expropriation costs of livestock reduction by . The simplistic approach to the economic dimension of the issue is thus deemed a limitation of the model, as it does not consider the wide range of economic implications. Further research should consider the wider economic impact of policy measures to gain a better understanding of economic dimension of the policy trade-off.

6.3.2 Legitimization of ecological engineering

Scientific backing of ecological engineering measures is required for ecological engineering to become a legitimate solution to the nitrogen crisis. The development of a detailed scientific knowledge base on the factors underlying N resilience of ecosystems is there for paramount. Such a knowledge base would allow for the design of effective and legitimate ecological policy measures. To this end, a detailed ecological understanding must be developed on the dynamic relation between nutrient availability and biodiversity. This knowledge base is essential as this dynamic relation underlies the N resilience of Natura 2000 areas. Such research should consider the relation between nutrient availability and different types of biota that exist in Dutch Natura 2000 areas. The research should investigate how nutrient availability impacts biota, and consequently results in a change of KDW. Only when this dynamic relation is well understood, can the effectiveness of ecological measures to increase the N resilience of Natura 2000 areas be determined.

For ecological measures to be put into practice as a N deposition mitigator, their potential for N resilience increasement must be quantified. The quantification of ecological measures to increase N resilience of ecosystems is critical to deduce the allowance of increased N depositions based on ecological compensation. Consequently, such research provides insights on the potential of ecological measures to relax emission guidelines and thereby mitigate the economic and societal impact of the nitrogen crisis. To this end, the ecological impact of a wide variety of ecological measures has to be empirically investigated based on field and lab experiments.

The empirical research on ecological measures should be focussed on mitigating the impact of nutrient availability. Nutrient availability mitigation refers to measures such as chalk or loam application that increase the buffer effect of the soil (Goulding, 2016). Such measures can be used to counter act the impact of nitrogen deposition as also mention in natura management plan of the Veluwe (Provincie Gelderland, 2017). Moreover, ecological measures can directly interfere with the state of biodiversity through the removal of undesirable plants and planting of desirable plants (Provincie Gelderland, 2017). Improving the state of biodiversity directly can thereby off-set the potentially negative impacts of increased nitrogen availability. The impact of such measures on the biota and the N resilience of Natura 2000 areas needs empirical substantiation. Also, ecological measures to extract polluting nutrients should be investigated. If ecological measures to extract nitrogen from soils can be empirically substantiated, without hurting the biota, it could provide an excellent compensatory measure to increased nitrogen depositions.

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Appendix A - Uncertainty analysis

A1 Ecology

A1.1 Uncertainty selection for ecological PRIM analysis

1. RealParameter("Foreign NH3 inflow", 30.16 , 44.64),
2. RealParameter("Foreign NOx inflow", 180.7, 271.08),
3. RealParameter("Initial N concentration soil", 0.1 , 0.15),
4. RealParameter("CG residence time N", 12.8, 19.2),
5. RealParameter("CG N background leaching factor", 176, 264),
6. RealParameter("Initial P concentration soil", 12 , 18),
7. RealParameter("CG residence time P", 11.2 , 16.8),
8. RealParameter("CG P background leaching factor", 200 , 300),
9. RealParameter("Uncertainty crop yield efficiency growth", 0.9 , 1),
10. RealParameter("Foreign water inflow", 70.2, 85.8),
11. RealParameter("N concentration foreign water", 2.28, 3.42),
12. RealParameter("P concentration foreign water", 0.088, 0.132),
13. RealParameter("Initial utilization of terrestrial biomass capacity", 0.7 , 0.9),
14. RealParameter("Initial share of nitrophilic biomass", 0.4, 0.6),
15. RealParameter("Other terrestrial biomass lifetime", 35, 45),
16. RealParameter("Nitrophilic biomass lifetime", 22.5, 27.5),
17. RealParameter("Other aquatic biomass lifetime", 7, 9),
18. RealParameter("Phosphoric biomass lifetime", 4.5, 5.5),
19. RealParameter("Initial utilization of aquatic biomass capacity AW", 0.7, 0.9),
20. RealParameter("Initial share of phosphoric biomass AW", 0.64, 0.96),
21. RealParameter("Initial utilization of aquatic biomass capacity RW", 0.56, 0.84),
22. RealParameter("Initial share of phosphoric biomass RW", 0.48, 0.72),
23. RealParameter("Initial utilization of aquatic biomass capacity LR", 0.16, 0.24),
24. RealParameter("Initial share of phosphoric biomass LR", 0.24, 0.36),
25. RealParameter("Terrestrial N:P function correction factor", 0.8, 1.2)
26. RealParameter("Aquatic N:P function correction factor", 0.77, 0.93)
27. RealParameter("Nitrogen impact correction factor", 0.9, 1.1),
28. RealParameter("Phosphorus impact correction factor", 0.9, 1),
29. RealParameter("Terrestrial biomass growth factor", 0.36 , 0.44),
30. RealParameter("Aquatic biomass growth factor", 1.8, 2.2),
31. RealParameter("Average driving distance diesel cars", 20293.2, 24802.8),
32. RealParameter("Average driving distance gas cars", 9787.5, 11962.5),
33. RealParameter("Effectiveness Dutch electric car policy", 0.5, 1),

A1.2 Comparison of model outcomes

All uncertainties

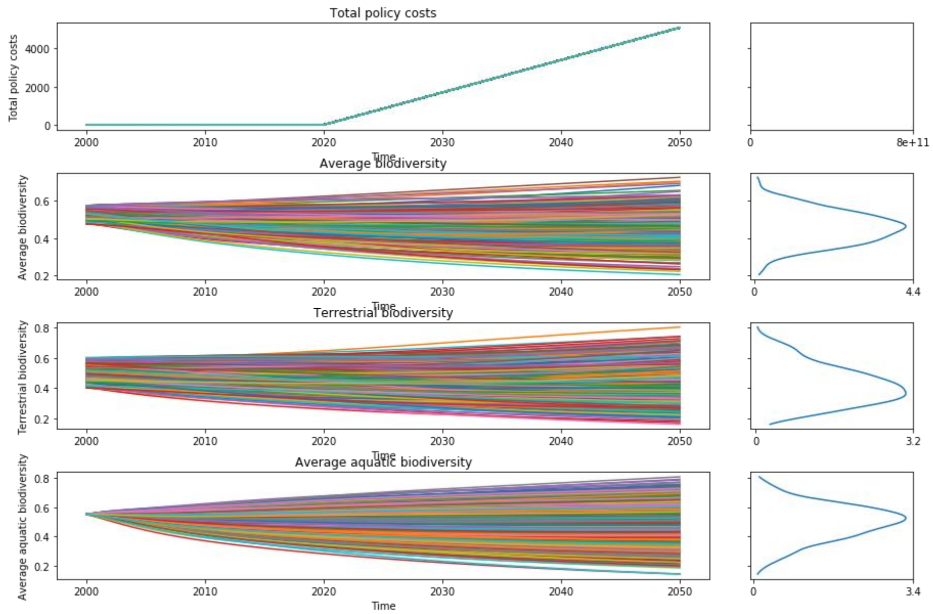


Figure A1: Model results when all uncertainties are active

Selection of uncertainties

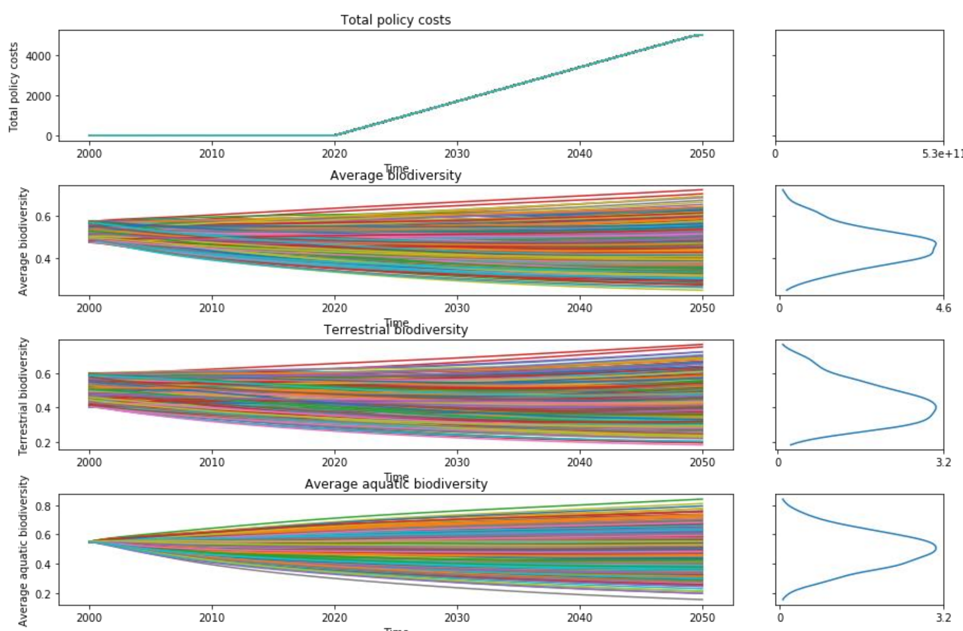


Figure A2: Model results for a selection of the uncertainties relevant to biodiversity

A1.3 PRIM results for ecological uncertainty analysis

The PRIM results show the uncertainties that together are predictive to certain outcomes of interests. The prim results are selected based on a density criteria of 80%. Indicating that that cases that are demarcated by the found uncertainty values, consist for 80% out of cases of interests. Correspondingly, a coverage of the cases of interests is found which represent the % of total outcomes of interest that is demarcated by the found uncertainties.

Table A1: PRIM results for cases of interests with an "Average biodiversity" score < 0.4

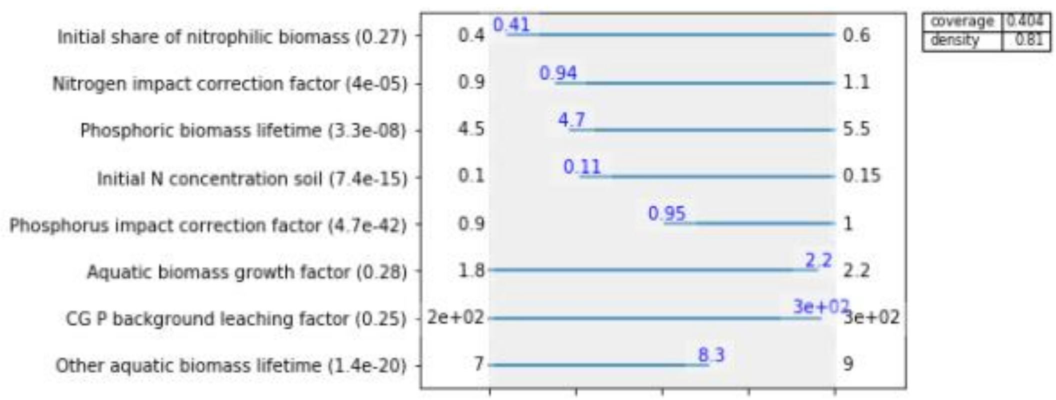


Table A2: PRIM results for cases of interests with an "Terrestrial biodiversity" score < 0.4

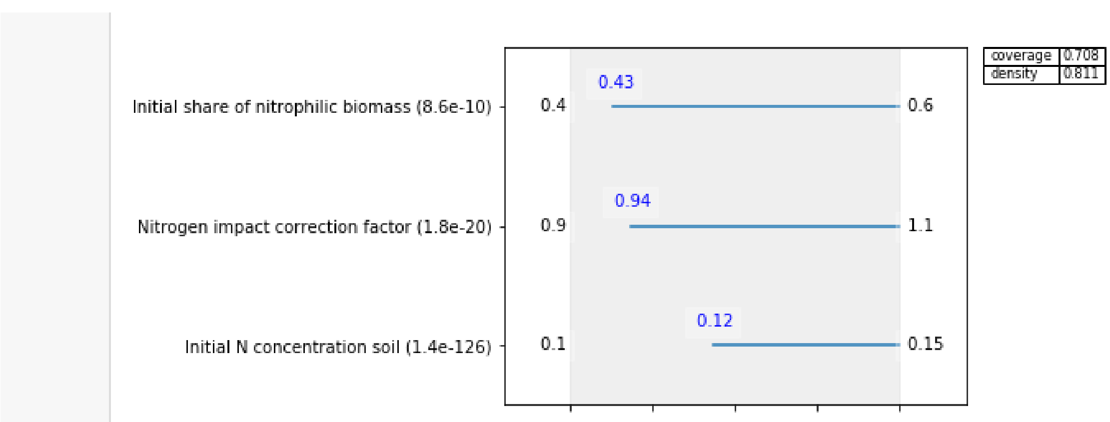
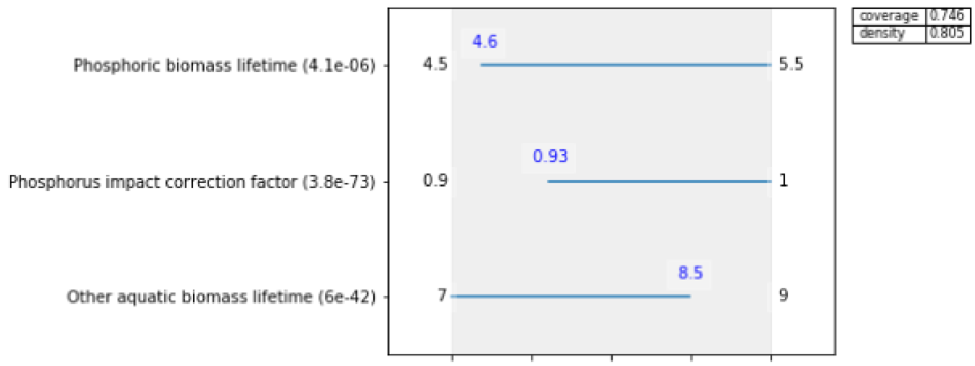


Table A3: PRIM results for cases of interests with an Aquatic biodiversity < 0.5



A2 EU guidelines

A2.1 Uncertainty selection for EU guidelines PRIM analysis

1. RealParameter("Initial N concentration soil", 0.1 , 0.15),
2. RealParameter("Denitrification rate NCG soils", 90 , 110),
3. RealParameter("Initial CG volatilization factor", 0.79 , 0.97),
4. RealParameter("CG denitrification rate", 0.11, 0.134),
5. RealParameter("Uncertainty crop yield efficiency growth", 0.9 , 1),
6. RealParameter("Initial utilization of terrestrial biomass capacity", 0.7 , 0.9),
7. RealParameter("Initial share of nitrophilic biomass", 0.4, 0.6),
8. RealParameter("Other terrestrial biomass lifetime", 35, 45),
9. RealParameter("Nitrophilic biomass lifetime", 22.5, 27.5),
10. RealParameter("Terrestrial N:P function correction factor", 0.8, 1.2),
11. RealParameter("N to P assimilation ratio", 11.25, 13.75),
12. RealParameter("Initial N:P availability ratio", 13.5, 16.5),
13. RealParameter("Nitrogen impact correction factor", 0.9, 1.1),
14. RealParameter("Terrestrial biomass to nitrogen conversion factor", 0.0008, 0.0012),
15. RealParameter("Terrestrial biomass growth factor", 0.36 , 0.44),
16. RealParameter("Cattle unit cow", 0.9, 1.1),
17. RealParameter("Cattle unit pig", 0.18, 0.22),
18. RealParameter("Cattle unit poultry", 0.00603, 0.00737),
19. RealParameter("Powerfeed N per cattle unit", 0.0000518, 0.0000632),
20. RealParameter("Roughfeed N per cattle unit", 0.00003587, 0.00004385),
21. RealParameter("Initial N in livestock", 640, 782),
22. RealParameter("Initial Manure N" , 453, 554.4),
23. RealParameter("Initial plant produce N", 315, 385),
24. RealParameter("Manure volatilization factor", 0.1018, 0.1244),
25. RealParameter("Livestock to manure N factor", 0.639, 0.781),
26. RealParameter("Average driving distance diesel cars", 20293.2, 24802.8),
27. RealParameter("Average driving distance gas cars", 9787.5, 11962.5),
28. RealParameter("Growth factor car demand", 1.0035, 1.0045),
29. RealParameter("Average car lifetime", 18, 22),
30. RealParameter("Initial amortization factor", 1.1, 1.5),
31. RealParameter("Uncertainty NOx shipping reduction trend (post 2020)", 1 , 1.2),
32. RealParameter("Uncertainty NOx industry reduction trend (post 2020)", 0.9 , 1.1),
33. RealParameter("Uncertainty NOx consumers, services, government and construction reduction trend (post 2020)", 1 , 1.15),
34. RealParameter("Uncertainty NOx agriculture and livestock trend (post 2020)", 1 , 1.2),
35. RealParameter("Uncertainty NH3 reduction trend (post 2020)", 0.8 , 1.2),
36. RealParameter("Effectiveness Dutch electric car policy", 0.5, 1),
37. RealParameter("Initial share of transitioned livestock farms", 0.105, 0.195),
38. RealParameter("Average farm transition cost", 0.8 , 1.2),
39. RealParameter("Potential for manure volatilization reduction", 0.5, 0.8),
40. CategoricalParameter("Livestock reduction cost scenario", (1 , 1.9)),

A2.2 Comparison of model outcomes

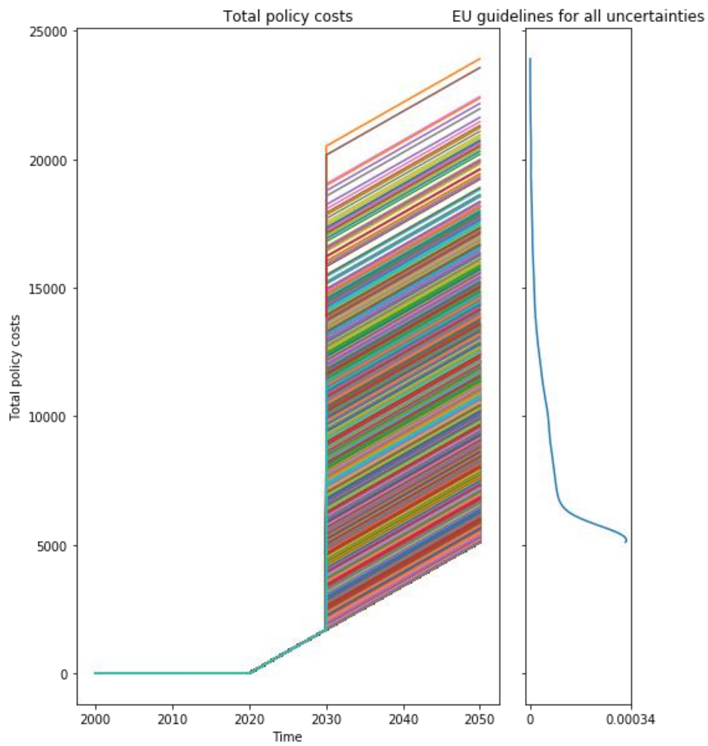


Figure A3: Total policy costs related to EU guidelines for all uncertainties

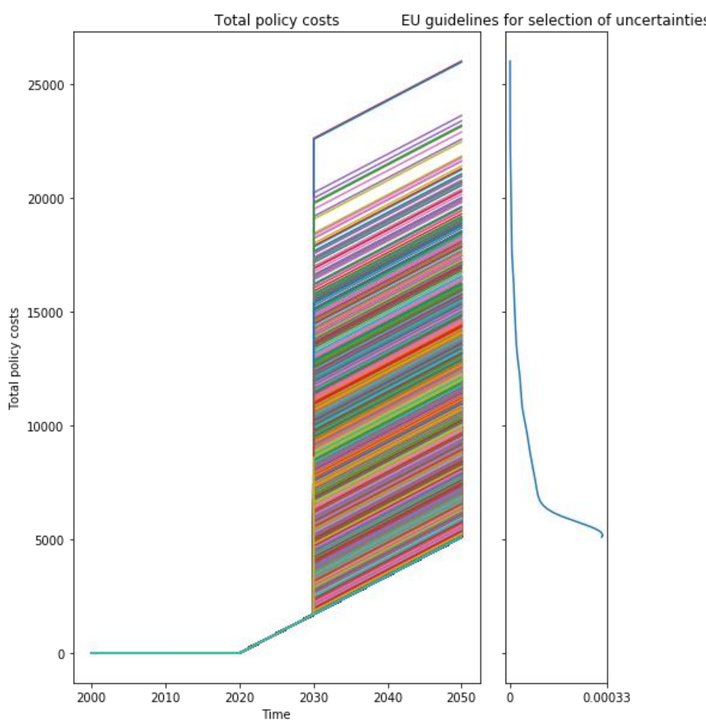
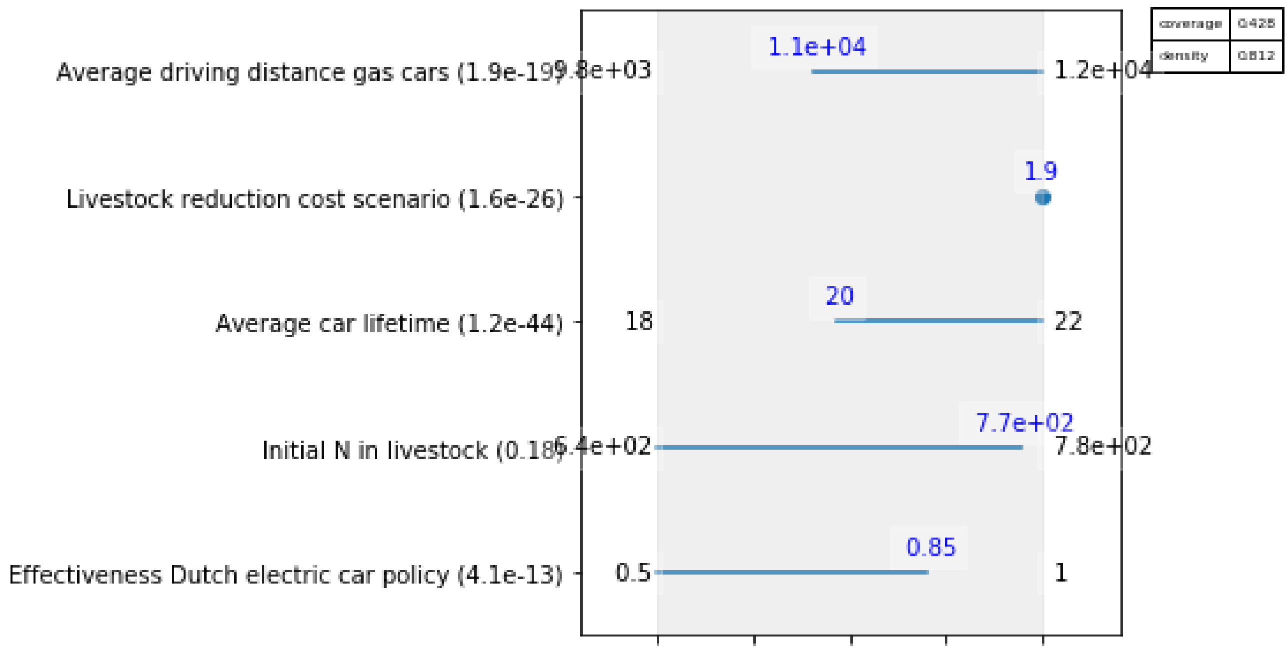


Figure A4: Total policy costs related to EU guidelines for a selection of uncertainties

A2.3 PRIM results for uncertainty analysis of EU guidelines

Table A4: PRIM results for cases of interests for EU guidelines for high cost scenarios



A3 NL guidelines

A3.1 Uncertainty selection for NL guidelines PRIM analysis

1. RealParameter("Initial N concentration soil", 0.1 , 0.15),
2. RealParameter("Denitrification rate NCG soils", 90 , 110),
3. RealParameter("Initial CG volatilization factor", 0.79 , 0.97)
4. RealParameter("CG denitrification rate", 0.11, 0.134),
5. RealParameter("Uncertainty crop yield efficiency growth", 0.9 , 1),
6. RealParameter("Initial utilization of terrestrial biomass capacity", 0.7 , 0.9),
7. RealParameter("Initial share of nitrophilic biomass", 0.4, 0.6),
8. RealParameter("Other terrestrial biomass lifetime", 35, 45),
9. RealParameter("Nitrophilic biomass lifetime", 22.5, 27.5),
10. RealParameter("Terrestrial N:P function correction factor", 0.8, 1.2)
11. RealParameter("N to P assimilation ratio", 11.25, 13.75),
12. RealParameter("Initial N:P availability ratio", 13.5, 16.5),
13. RealParameter("Nitrogen impact correction factor", 0.9, 1.1), #less sensitive
14. RealParameter("Terrestrial biomass to nitrogen conversion factor", 0.0008, 0.0012),
15. RealParameter("Terrestrial biomass growth factor", 0.36 , 0.44),
16. RealParameter("Cattle unit cow", 0.9, 1.1),
17. RealParameter("Cattle unit pig", 0.18, 0.22),
18. RealParameter("Cattle unit poultry", 0.00603, 0.00737),
19. RealParameter("Powerfeed N per cattle unit", 0.0000518, 0.0000632),
20. RealParameter("Roughfeed N per cattle unit", 0.00003587, 0.00004385),
21. RealParameter("Initial N in livestock", 640, 782),
22. RealParameter("Initial Manure N" , 453, 554.4),
23. RealParameter("Initial plant produce N", 315, 385),
24. RealParameter("Manure volatilization factor", 0.1018, 0.1244),
25. RealParameter("Livestock to manure N factor", 0.639, 0.781),
26. RealParameter("Average driving distance diesel cars", 20293.2, 24802.8),
27. RealParameter("Average driving distance gas cars", 9787.5, 11962.5),
28. RealParameter("Growth factor car demand", 1.0035, 1.0045),
29. RealParameter("Average car lifetime", 18, 22),
30. RealParameter("Initial amortization factor", 0.1, 1.5),
31. RealParameter("Uncertainty NOx shipping reduction trend (post 2020)", 1 , 1.2),
32. RealParameter("Uncertainty NOx industry reduction trend (post 2020)", 0.9 , 1.1),
33. RealParameter("Uncertainty NOx consumers, services, government and construction reduction trend (post 2020)", 1 , 1.15),
34. RealParameter("Uncertainty NOx agriculture and livestock trend (post 2020)", 1 , 1.2),
35. RealParameter("Uncertainty NH3 reduction trend (post 2020)", 0.8 , 1.2),
36. RealParameter("Maximum turfing norm", 0.4, 0.6),
37. RealParameter("Turfing cost per km2", 0.8, 1.2),
38. RealParameter("Turfing effectiveness for nutrient removal", 0.56, 0.84),
39. RealParameter("Landscaping effectiveness", 0.35, 0.65),
40. RealParameter("Area per gardener", 0.288, 0.432),
41. RealParameter("Cost per gardener", 0.08, 0.12),
42. RealParameter("Effectiveness Dutch electric car policy", 0.5, 1),
43. RealParameter("Initial share of transitioned livestock farms", 0.105, 0.195),
44. RealParameter("Average farm transition cost", 0.8 , 1.2),
45. RealParameter("Potential for manure volatilization reduction", 0.5, 0.8),
46. CategoricalParameter("Livestock reduction cost scenario", (1 , 1.9)),

A3.2 Comparison of model outcomes

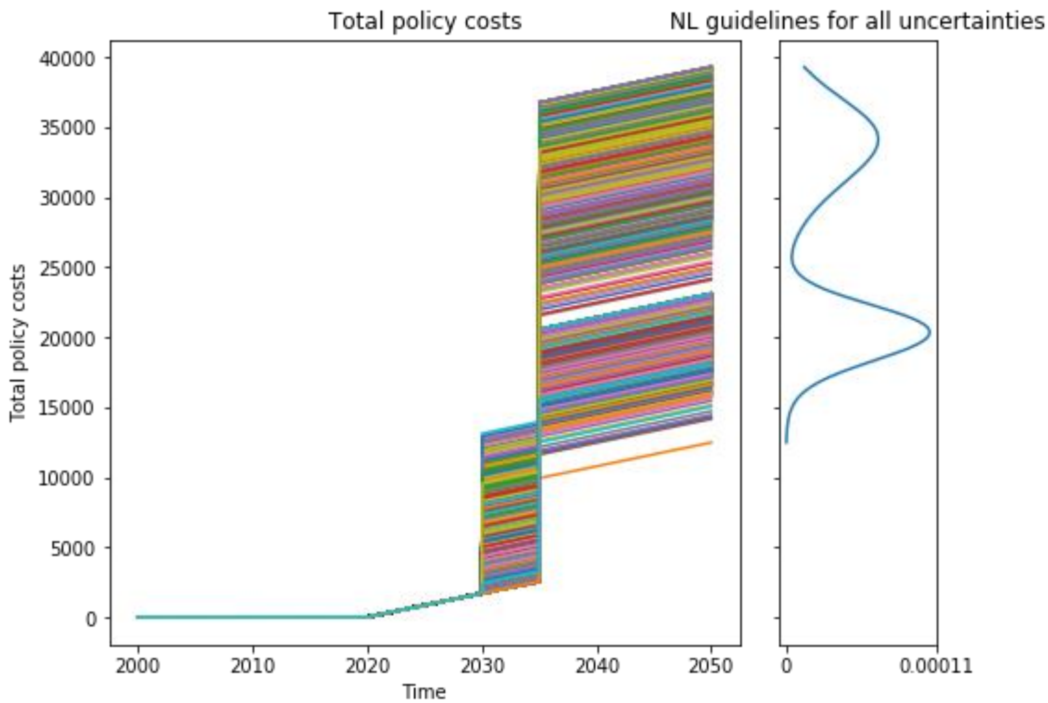


Figure A5: Total policy costs related to NL guidelines for all uncertainties

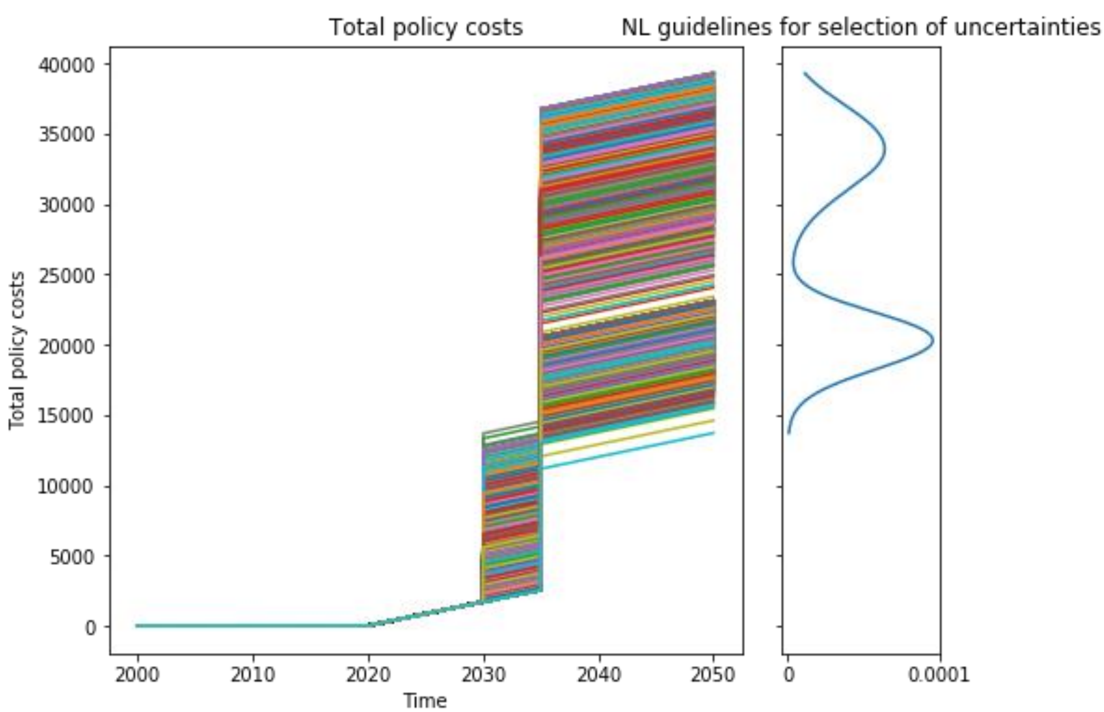
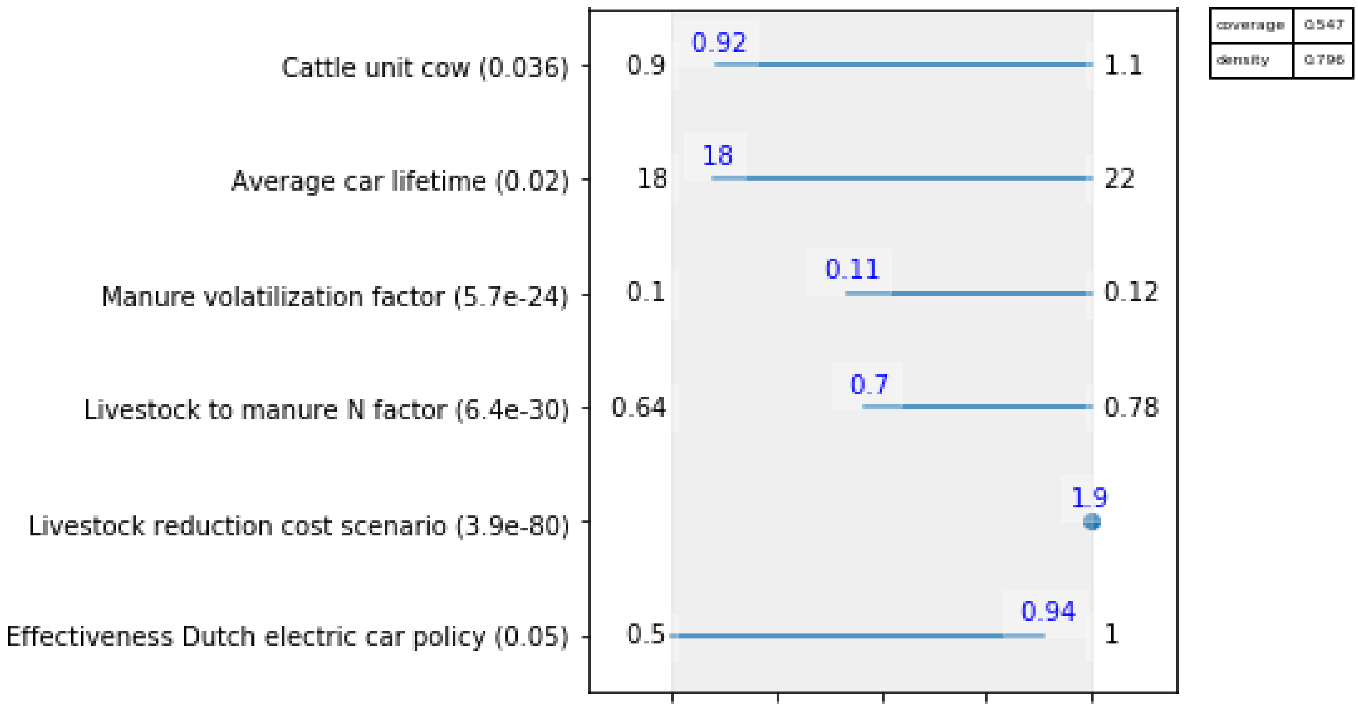


Figure A6: Total policy costs related to NL guidelines for a selection of uncertainties

A3.3 PRIM results for uncertainty analysis of NL guidelines

Table A5: PRIM results for cases of interests for NL guidelines for high cost scenarios



Appendix B – Ecological results

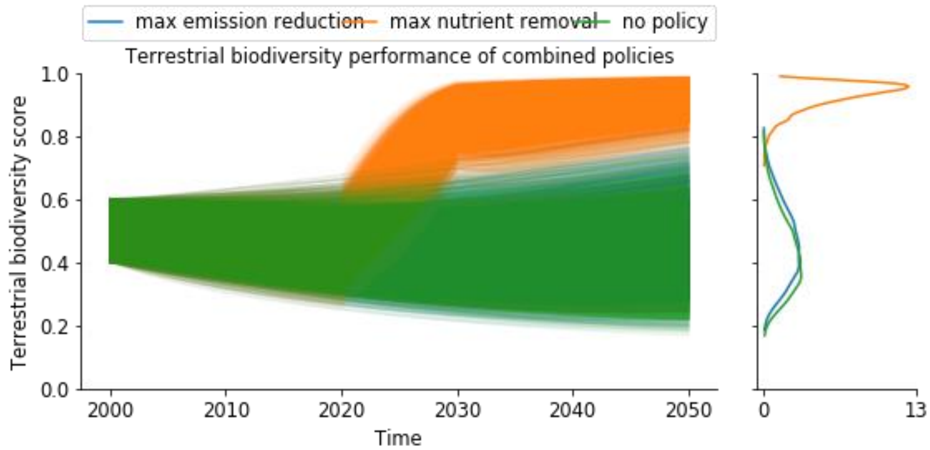


Figure B1: Terrestrial biodiversity performance of combined policies

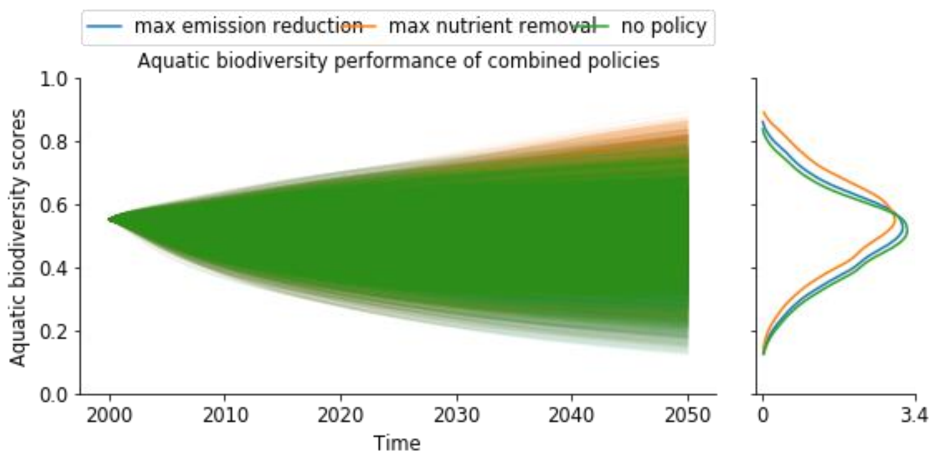


Figure B2: Aquatic biodiversity performance of combined policies

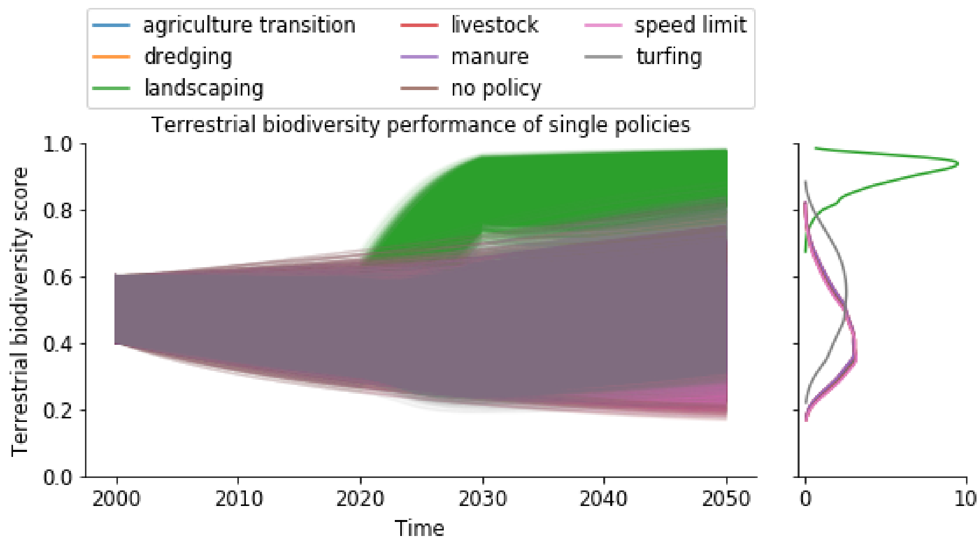


Figure B3: Terrestrial biodiversity performance of single policies

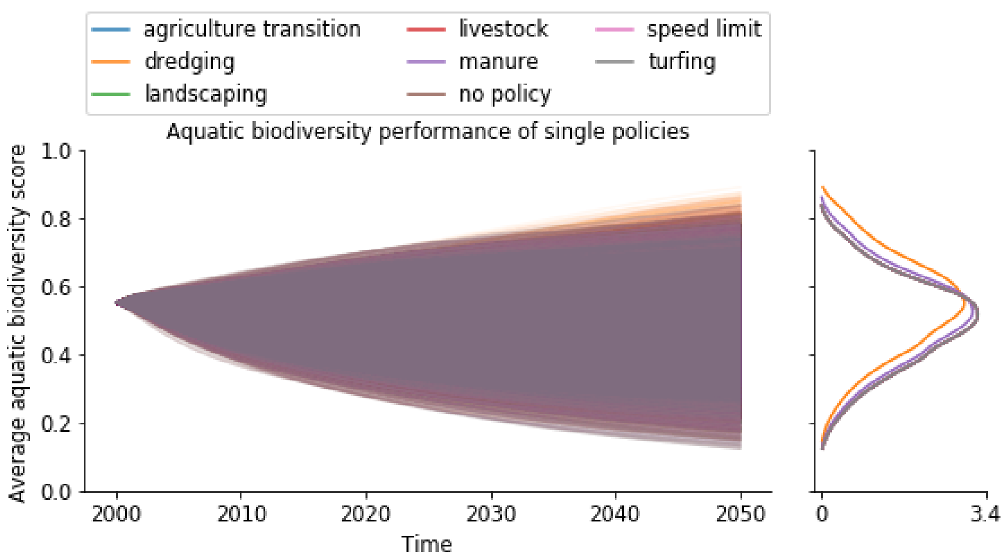


Figure B2: Aquatic biodiversity performance of single policies

Appendix C - EU guidelines results

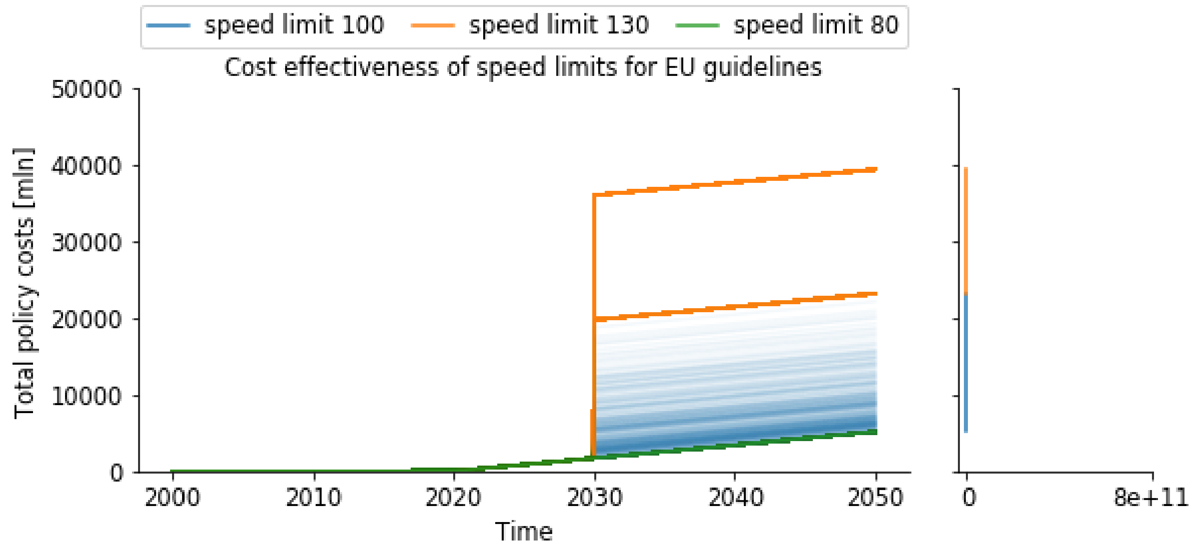


Figure C1: Cost effectiveness of speed limits for EU guidelines

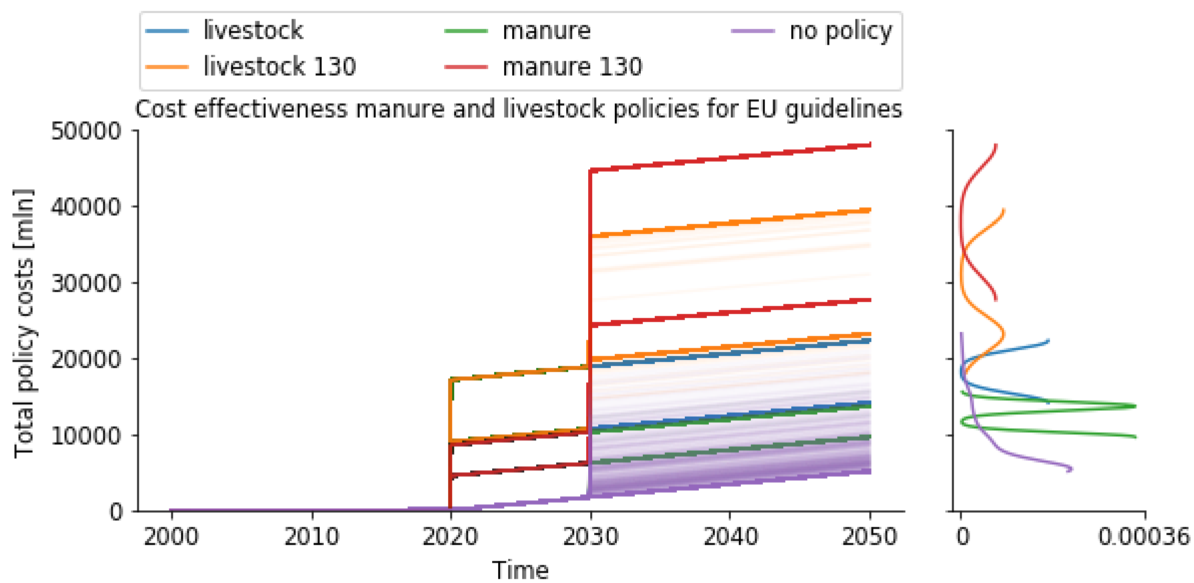


Figure C2: Cost effectiveness manure and livestock policies to meet EU guidelines in combination with a speed limit of 130 km/h

Appendix D - NL guidelines results

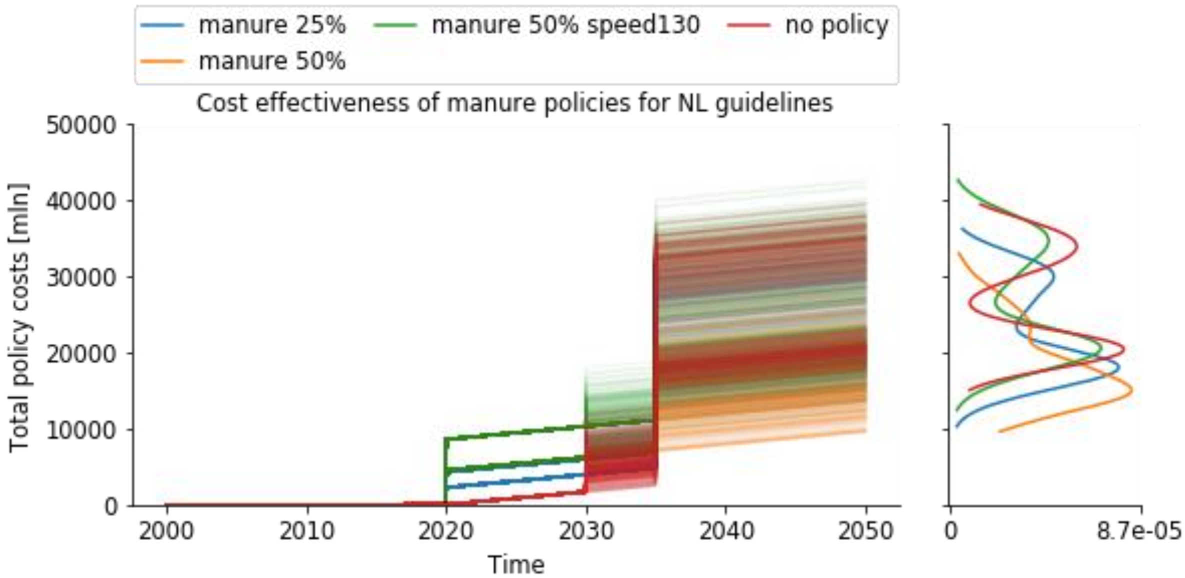


Figure D1: Cost effectiveness of manure policies for NL guidelines

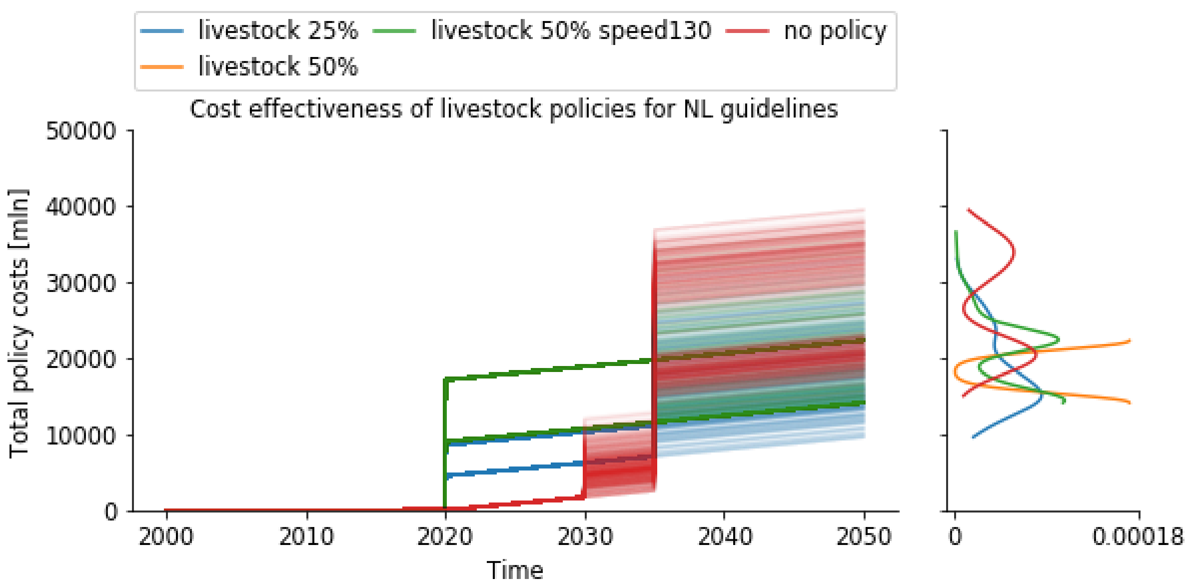


Figure D2: Cost effectiveness of livestock policies for NL guidelines

Appendix E - Adaptive NL guidelines

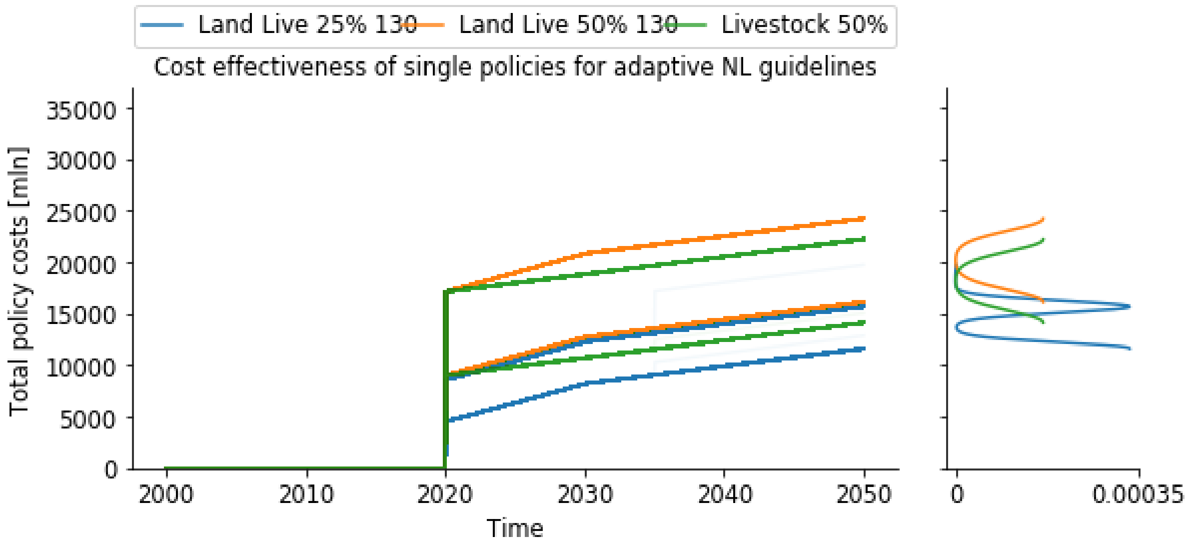


Figure E1: Cost effectiveness of livestock policies in combination with landscaping and speed limit of 130 km/h for adaptive NL guidelines

Appendix F - Atmosphere

The amount of reactive nitrogen (Nr) in the atmosphere is central to the Dutch system of nutrient pollution. The atmosphere is the medium through which nutrient pollutants in the form of ammonia (NH_3) and nitrogen oxides (NO_x) are distributed. The amount of Nr in the atmosphere changes based on the in- and out-flows of Nr. The in-flow of pollutants stem from either national or transboundary sources. After being taken up into the atmosphere the pollutants can either deposit or be exported outside the Dutch borders, thereby reducing the available pool of Nr. The data used for the quantification of the parameters is based on the year 2017.

The initial value of Nr in the atmosphere is determined based on NH_3 and NO_x concentrations from data provided by the CLO database (RIVM, 2013, 2020a). By multiplying these concentrations with the volume of the Dutch atmosphere, the total weight of NH_3 and NO_x available in the atmosphere is calculated. The initial total amount of Nr in the atmosphere is then estimated based on the share of Nr in NO_x (30.57%) and NH3 (82.35%) (TNO, 2019), equaling 40 kilotons.

The amount of Nr inflow and outflow in the Dutch atmosphere are determined based on domestic and foreign flows. The inflow of Nr into the Dutch atmosphere is sourced from either domestic Nr emissions or transboundary Nr inflow. The outflow of Nr from the Dutch atmosphere occurs either through domestic deposition or through transboundary outflow. To determine the value for the Nr in-and out-flows data is used as reference. Domestic measuring systems (LMM) and models (EMEP) provide clear data on the amount of depositions and domestic emissions. However, specific data on the amount of transboundary emission flows, especially foreign inflows, is not presented by the authorities. The determination of transboundary inflow and outflow are therefore based on the assumption that depositions rates of NO_x and NH3 emissions are equal for foreign and domestic emissions. Based on this assumption the size of transboundary NO_x and NH_3 flows is deduced from data which relates depositions to either foreign or domestic emissions.

F1 Atmospheric inflows

The atmospheric inflows occur through domestic NO_x and NH_3 emissions and transboundary NO_x and NH_3 inflows.

Domestic Nr Emissions

The inflow of Domestic Nr emissions is determined based on the share of Nr in domestic emissions of NH_3 and NO_x. Domestic Nr emissions occur from domestic emission sources of NO_x or NH_3. The vast majority of NH_3 emissions originate from the Dutch agricultural industry (Adviescollege Stikstofproblematiek, 2020;

TNO, 2019). Whilst NO_x emissions originate from combustion processes which occur in a large variety of Dutch sectors such as; industries, traffic and consumer behaviour (Adviescollege Stikstofproblematiek, 2020; TNO, 2019).

To model the domestic NO_x emissions a demarcation is made between traffic sources, agriculture sources, industry and consumer & services related emissions. The demarcation is made based on data provided by the nitrogen advice committee (Adviescollege Stikstofproblematiek, 2020). Additionally, trends for future reductions are added to model the development of emissions into the future. This is done for NO_x sources that are not related to traffic and the NH_3 sources except those related to agriculture. The trends are deduced from previous trend as provided by the CLO (RIVM, 2013, 2019) and estimations by the nitrogen advice committee (Adviescollege Stikstofproblematiek, 2020)

The NO_x emissions from agriculture are modelled based on the relative size of the sector. This is done as NO_x emissions from agriculture originate from the use of heavy machinery. The assumption is made that the use of heavy machinery can be estimated relative to the size of the industry. Thus the larger the size of the agriculture sector, the more use of heavy agriculture machinery. The total amount of plant produce is used as a proxy for the size of the agriculture sector.

The NO_x emissions from traffic are estimated based on the composition of the car fleet, where a distinction is made between gasoline cars, diesel cars and electric cars. With the transition goals of the Dutch government will change the composition of the fleet to be increasingly more electric. A sub model is constructed to model the transition of car use based on the yearly car demand and the choice factor for electric cars which is influenced by government policy. From the fleet composition, the average driving per car, and average NO_x emissions per km the total amount of NO_x from traffic is modelled.

Foreign Nr inflow

Foreign Nr inflow is calculated based on the share of Nr from the inflow of foreign NH_3 and NO_x emissions. The amount of NO_x and NH_3 inflow is based on data from the CLO (RIVM, 2019) and TNO (TNO, 2019). No specific data is available on the amount of transboundary NO_x and NH_3 inflow into the Dutch atmosphere. Therefore, these flows have to be deduced from other available information on depositions occurring from foreign emissions. The CLO provides data on depositions of NO_x and NH_3, and the share of which occurs from foreign emissions. By assuming that the deposition factor of domestic and foreign emissions are equal, the amount of foreign NO_x and NH_3 inflow can be determined. First, the domestic deposition factor is determined from TNO data on domestic emissions and CLO data on domestic depositions. By assuming that the foreign deposition factors of NO_x and NH_3 are equal to their respective domestic deposition factors, the foreign inflow of NO_x and NH_3 is determined from CLO data on foreign depositions of NH_3 and NO_x and the deposition factor of NH_3 and NO_x.

Table 1: Overview of initial foreign and domestic emissions, depositions, and corresponding deposition and outflow factors.

	Emissions (kt)	Nr emissions (ktn)	Depositions (ktn)	Deposition factor (% of emissions in Nr)	Transboundary outflow (ktn)	Outflow factor
Total						
Nr	283,13 (Total Nr from NO_x + Nr from NH_3)	283,13 (Total Nr from NO_x + Nr from NH_3)	96,282 (CLO 2)	35,13% (= domestic depo factor)	**186,848**	**65,99%**
NO_x	468 (Conv. Fact = 3,271)	143,07 (Foreign + Domestic)	30,81 (32 % of total Nr depo) (CLO 1)	21,47% (= domestic NO_x depo factor)	**112,26**	**78,47%**
NH_3	170,07 (Conv. Fact = 1,214)	140,06 (Foreign + Domestic)	65,47 (68 % of total Nr depo) (CLO 1)	46,81% (= domestic NH3 depo factor)	**74,59**	**53,26%**
Domestic						
Nr	183 (TNO)	183 (TNO)	66,9 (69,5% of total Nr depo) (CLO 1)	36,55%	**116,1**	**63,44%**
NO_x	242 (TNO)	74 (Conv. Fact. = 0,3057) (TNO)	15,89 (16,5% of total Nr depo) (CLO 1)	21,47%	**58,11**	**78,53%**
NH_3	132 (TNO)	109 (Conv. Fact. = 0,8235) (TNO)	51,03 (53% of total Nr depo) (CLO 1)	46,81%	**57,97**	**53,18%**
Foreign						
Nr	100,13 (Foreign NO_x + Foreign NH_3)	100,13 (Foreign NO_x + Foreign NH_3)	29,36 (30,5% of total Nr depo) (CLO 1)	29,32%	**70,77**	**70,68%**
NO_x	225,9 (Conv. Fact = 3,271)	69,07 (= depo * (1/ NO_x depo fact.)	14,83 (15,4% of total Nr depo) (CLO 1)	21,47% (= domestic NO_x depo factor)	**54,24**	**78,53%**
NH_3	37,7 (Conv. Fact = 1,214)	31,06 (= depo * (1/ NH_3 depo fact.)	14,54 (15,1% of total Nr depo) (CLO 1)	46,81% (= domestic NH3 depo factor)	**16,52**	**53,19%**

**Legend – Blue cells are input data the orange and grey cells are deduced.*

F2 Atmospheric outflows

Nr Depositions

Depositions occur when Nr in the shape of NO_x or NH_3 transition from the atmosphere to the soil. The amount of depositions are relative to the amount of NO_x and NH3 emissions, either from domestic or foreign sources. The deposition factors of NH_3 (46.81%) and NO_x (21.47%) differ due to the longer emission trajectory of NO_x compared to NH_3. NH_3 thus deposits closer to the emission source, and has a larger deposition factor on domestic soils, compared to NO_x. Contrarily, NO_x has a lower deposition factor, meaning a larger portion of NO_x emissions do not deposit and instead transfer to transboundary atmospheres.

The domestic depositions are assumed to be distributed uniformly over the domestic surface. This assumption neglects the non-uniform emission distribution of NO_x and NH_3 depositions from its emission source. As the implementation of geographical distributions is not feasible within the confines of this study, the assumption of uniform deposition is necessary. Due to the assumption of uniform distribution of depositions, the relative impact of high emission regions close to Natura 2000 areas are underestimated, which should be considered in the model results. Based on this assumption the deposition of Nr to waterbodies, culture ground or non-culture ground can be modelled relatively to their respective surface area.

Transboundary Nr outflow

A significant portion of Nr emissions, which is either domestically emitted or originates from foreign sources, flows out of the Dutch atmosphere. These flows are referred to as "transboundary Nr outflows", and occur due to the long emission trajectories of Nr compounds. The amount of transboundary outflow is modelled as the proportion of emissions that does not deposit. This reasoning assumes that the chemical reactions of NH_3 and NO_x in the atmosphere is negligible. The outflow factors of NH_3 and NO_x are thus determined based on their respective deposition factors (e.g. Outflow factor = 1 – Deposition factor). By multiplying the total emissions of NH_3 and NO_x with their outflow factor, the total amount of Transboundary Nr outflow is determined.

Appendix G – Soils

Soils have a central role in the issue of nutrient pollution as they function as an important medium for the distribution of nutrients to plants, but also to other mediums such as water and air. Additionally, soils directly impact the state of terrestrial biodiversity, as the accumulation of nutrients in soils provide the key building blocks underlying plant growth (Chapin III et al., 2011). To determine the impact of nutrient pollution on the state of biodiversity it is therefore critical to adequately model soil nutrient dynamics. As soil nutrient dynamics are strongly dependent on anthropogenic practices, a categorization of soil types is made.

To model the distribution of nutrients and the impact of nutrient pollution on biodiversity, the model distinguishes between culture ground stocks and non-culture ground stocks. This modelling choice is made as culture grounds and non-culture grounds are subject to vastly different influences. Culture grounds refer to agricultural lands which are heavily loaded with nutrient rich manures and fertilizers. The heavy loading of these nutrients requires a separate analysis due to the strong impact it has on the nutrient dynamics in the soil and the consequent emissions to air and water. Non-culture ground soils are not subject to agricultural practices, meaning the soil nutrient dynamics are less impacted by anthropogenic influences and proceed in large according to natural processes.

Within the non-culture ground soils, a distinction is made between Natura 2000 areas, which represent the areas belonging to the Natura 2000 network (Rijkswaterstaat, 2021), and other non-culture ground soils. The distinction is made because Natura 2000 areas are of critical importance to the performance of nature preservation policy, as these must be protected according to European preservation guidelines. The distinction allows for the analysis of the state of biodiversity in Natura 2000 areas and thus the success of policies for the mitigation of nutrient pollution and nature preservation. The other non-culture ground soils are still incorporated in the model as they still facilitate the significant process of denitrification which is relevant to the policy issue.

G1 Nutrient flows through culture grounds

Culture grounds are used by farmers for the cultivation of crops and the grazing of cattle. These agricultural practices result in significant changes of the flow of nutrients, due to the application of manure and fertilizers (CBS, 2020b). These changes are noticeable in the amount of nutrient pollution that occurs to waterbodies due to processes of leaching and run-off (Chardon & Schoumans, 2002; Schoumans, 2015). Moreover, the high availability of nitrogen as a results of application practices increases the amount of denitrification and volatilization of nitrogen to the atmosphere. And lastly, application practices increase the amount of crop yield, which thereby strongly impacts the outflow of nutrients in the form of plants.

Aquatic emission pathway

The addition of manure and fertilizers to the soils strongly impacts the amount of nutrient pollution to water (RIVM, 2020c), as the high availability of N and P in the soil increases the amount of leaching and run-off of nutrients to waterbodies (Oenema et al., 2005). Nutrient pollution to water in culture grounds occur in two forms, namely, directly in the form of leaching and run-off, and indirectly through background leaching (Oenema et al., 2005; Withers & Haygarth, 2007). Direct leaching occurs as a consequence of meteorological factors such as rain which wash out nutrients from the soil into nearby waterbodies. The amount of direct leaching which occurs is dependent on the nutrient availability in the soil. Direct leaching is therefore modelled based on the availability of nutrients in the soil and an estimated residency time which aligns with data on nutrient concentrations found in waterbodies (RIVM, 2020c).

Contrarily to direct leaching, background leaching occurs slowly overtime as a consequence of slow seeping of nutrients deeper into the soil, where they then become available to open waters over time (Oenema et al., 2005; Withers & Haygarth, 2007). This form of background leaching is found to be significant in culture grounds, presumable because nutrient surpluses accumulate in the subsoils over time. The amount of background leaching is assumed to stay constant over time. This assumption is made due to a lack of insight on deep soil nutrient dynamics and its long-term impact on background leaching.

Arial pathway

The application of manure and fertilizers results in a high availability of nitrogen in the soil which increases the amount of nitrogen which is lost to the atmosphere, through processes of denitrification and volatilization. The amount of arial loss of nitrogen is determined based on CBS data on the flow of nitrogen (CBS, 2020b). The arial outflow of nitrogen is modelled as a percentage of the total reactive nitrogen (Nr) which flows into the culture ground. This modelling approach is deemed appropriate as the arial loss of nitrogen happens shortly after application. Making the arial outflow directly dependent on the inflow is therefore deemed the more appropriate approach, compared to making the outflow stock dependent.

The arial outflow of phosphor from the soil is not incorporated in the model as literature and data suggests that the arial emission pathway is not significant for phosphates (Berhe et al., 2010; CBS, 2020b). Additionally, the arial pathway of phosphate emissions only occurs over very short distances in the form of dust, meaning that the likelihood of pollution to Natura 2000 areas is minimal. Moreover, arial pollution of phosphates is heavily dependent on meteorological circumstances and application practices, as it only occurs in the form of dust. Based on the combination of insignificance and high dependency on complex variables the arial pathway of phosphates is not incorporated in the model.

Cropyield

With the goal of agriculture practices being the cultivation of crops, a majority of the nutrient flows occur in the form of plants (CBS, 2020b). The crop yield that occurs from farming practices is heavily dependent on the availability of phosphor and nitrogen as these are fundamental nutrients for the assimilation of biomass. With culture grounds being typically poor in N and P nutrients, the dependency of the harvest on manure and fertilizer application is exacerbated. To this end the harvesting of crop is modelled based on the total amount of applied nutrients.

G2 Nutrient flows through non-culture grounds

Opposed to culture grounds, non-culture grounds are not subjective to direct anthropogenic practices. Instead, the soil dynamic processes occur in large through natural processes. The main natural processes which occur in non-culture grounds are related to denitrification and the accumulation and decomposition of biomass. The aquatic pathway is not directly incorporated as data on the leaching from non-culture grounds soils is not readily available. Instead, the aquatic nutrient pollution pathway from non-culture grounds is incorporated indirectly through an inflow of nutrients by means of rain in the regional water stock.

As mentioned, a distinction is made between the Natura 2000 areas and other NCG areas. The Natura 2000 areas are modelled in detail due to its relevance for the measuring of nature preservation. On the contrary, the state of biodiversity in “Other NCG” stock is not relevant to the policy issue of nature preservation. Only the natural process of denitrification which occurs in “Other NCG” is relevant to the issue of nutrient. Additionally, the composition and decomposition of biomass in the “other NCG” stock does not impact the other stocks as the aquatic pathway is incorporated separately. For these reasons, state of biomass and biodiversity in the “Other NCG” stock is not relevant to the policy issue. This stock is therefore not expanded on beyond its process of denitrification.

Arial emission pathway

The arial transportation of nitrogen occurs in non-culture grounds through the natural process of denitrification. For both non-culture ground stocks this process is incorporated based on a natural base rate of denitrification per area. By multiplying the area of each of the non-culture ground stocks, the total amount of Nr from denitrification from these areas is determined.

Biomass pathway

Nutrients flow in and out of Natura 2000 soils through processes of biomass assimilation and decomposition. The availability of phosphorus and nitrogen impact the growth of biomass in these areas as these nutrients are the fundamental building blocks underlying plant growth (Chapin III et al., 2000). Based on the availability of these nutrients and the state of biomass in the area, the in and outflow of nutrients is determined. The availability of N and P nutrients impacts the assimilation of biomass through two processes: nitrogen availability and nutrient limitation.

The availability of nitrogen impacts biodiversity in terrestrial ecosystems as most plants perform best in nitrogen poor environments (Bouwman et al., 2002). To limit the modelling complexity, the assumption is made that all terrestrial Nature 2000 areas are nitrogen limited. This assumption aligns with what the literature suggests, namely that most terrestrial ecosystems are nitrogen limited (P. Vitousek & Field, 2001). In the nitrogen limited areas, the high availability of nitrogen is conducive to the growth nitrophilic biomass. Non-nitrophilic biomass is thereby at a competitive disadvantage in nitrogen rich areas, which makes the high availability of nitrogen damaging to the state of biodiversity. This dynamic is incorporated in the model with the use of the "Nitrogen impact" variable, which represents the impact of nitrogen availability on nitrophilic plant growth based on the "nitrogen impact on growth function". This function is constructed to represent a reducing trend of nitrogen impact when availability decreases. To ensure viable model behaviour and the possibility of uncertainty testing a "nitrogen impact correction factor" is also incorporated.

In addition, nutrient limitation impacts the growth of biomass based on the relative availability of nutrients. As biomass requires both N and P nutrients for growth, limitation of either one reduces the amount of potential biomass growth. This dynamic is incorporated in the terrestrial biodiversity sub-model with the "Terrestrial N:P ratio function", which is based on the assumption that terrestrial ecosystems are nitrogen limited. With this function the impact of nutrient limitation on growth is determined based on the N:P ratio. The function itself is derived from literature data on the impact of nutrient limitation on plant growth. It describes the relation where low N:P ratios indicate high degrees of N limitation and thus lower overall biomass growth. Conversely, high N:P ratios indicate low degrees of N limitation and thus high utilization of the biomass growth potential.

Appendix H – Water

Water is an important medium for biodiversity which is heavily impacted by nutrient pollution. Nutrients are distributed throughout aquatic ecosystems by the flow of water. Nutrient pollution in water can result in eutrophication and thereby the loss of biodiversity. International water guidelines such as the KRW are set in place to protect water bodies from excessive nutrient availability. These preservation measures are crucial as large parts of the Nature 2000 areas encompass aquatic ecosystems (Rijkswaterstaat, 2021), which therefore must be protected.

To model the state of the biodiversity in aquatic ecosystems and the impact of preservation policies on biodiversity, the flow of nutrients throughout the Dutch river delta is modelled. This is done based on a simple model for the flow of water throughout the Dutch river delta which functions as a medium for the distribution of nutrients throughout the different waterbodies. Additionally, the inflow of nutrients to the Dutch river delta is modelled based on transboundary inflow and domestic pollution sources. From the flow of nutrients throughout the waterbodies the impact of nutrient pollution on aquatic biodiversity is estimated.

H1 Water flow

The Dutch water system can be characterized as a river delta, meaning that all the water that flows in also flows out. Assuming that groundwater levels stay constant over time, the flow of water and nutrients through the Dutch river delta can thereby be modelled based on the transboundary and meteorological inflow of water. To this end, CLO data on the nutrient concentration in waterbodies, the inflow of nutrients from industrial and agricultural activities, and the waterflow throughout the different waterbodies is used. The CLO data on nutrient concentrations demarcates three types of waterbodies, namely; agricultural water bodies, regional water bodies and "Rijkswateren". The Rijkswateren encompass the largest water bodies in the Netherlands which originate from the river Rhine and Maas (Rijkswaterstaat, 2021).

To model the nutrient flows throughout the waterbodies, the nutrient concentrations in the waterbodies have to be linked to the flow of water throughout the respective water bodies. CLO data on the waterflow from transboundary inflow and meteorological factors is used to formulate a simple model for the waterflows throughout the Dutch river delta (RIVM, 2003). Assumptions on the flow of water throughout the Dutch river delta must be made to match the waterflow data with the water concentrations in the demarcated waterbodies;

1. Waterflow between waterbodies is assumed to occur from Agricultural water, to regional water to Rijkswateren. This assumption is based on the notion that the Dutch water system functions as a river delta where all the water that flows in must flow out. The regional waters are hereby identified as the medium by which water is transported from the smallest waterbodies to the Rijkswateren which facilitate the outflow of water from the Dutch water system.
2. Metreological water inflows occur through the Agricultural and Regional water body. The assumption is based on the notion that rain is distributed uniformly over the Dutch lands, meaning that they are most likely to firstly be transported into the smaller waterbodies before being joined in the larger Rijkswateren. The waterflow from rain is thereby distributed over the agricultural and regional waterbody based on their respective surface area. The assumption is limited as it neglects the impact of sewage systems on the waterflow.
3. Evaporation is assumed to occur proportionally to the surface area of the respective waterbodies. The data on evaporation waterflows is thereby distributed based on the relative surface area of the waterbodies.
4. All transboundary water inflows occur through the Rijkswateren. With the Rijn, Maas being Rijkswateren and accounting for ~95% (RIVM, 2003) of the transboundary water inflow, this assumption is deemed viable.

From these assumptions the following simple model for the flow of water in the Netherlands is setup (Fig. H1). Based on the model for water flow and information on water concentrations in the respective waterbodies, the total volume of nutrients which flows throughout is delta is modelled. The modelling of nutrients is done separately for N and P. Meaning, two parallel systems of nutrients flows are setup, based on the same model for water flows.

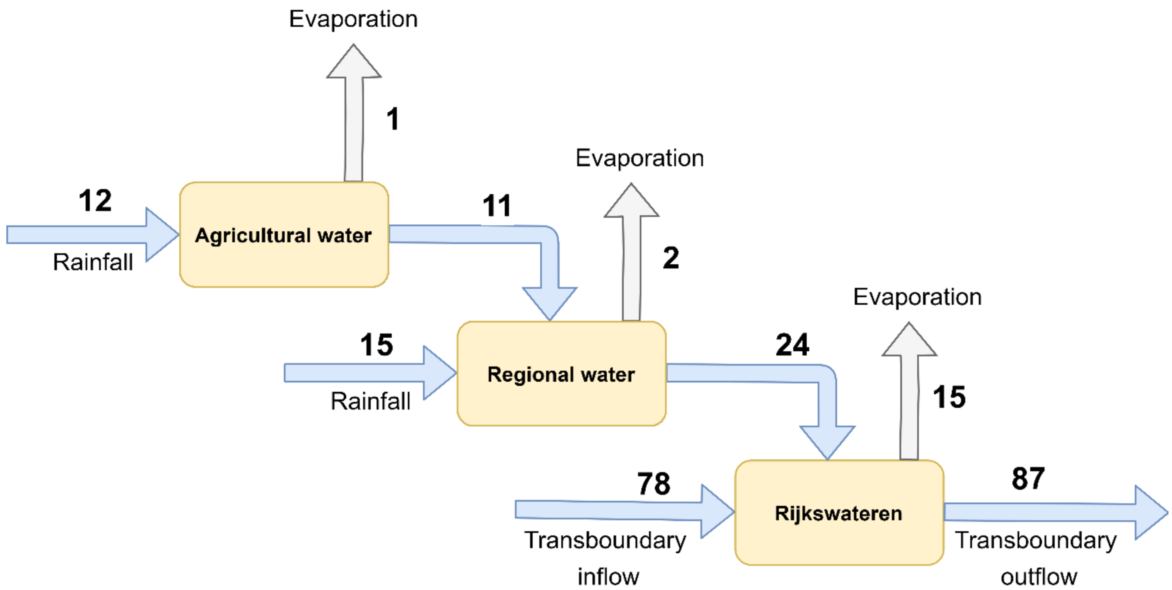


Fig H1: Simple model for waterflow throughout the Dutch river delta.

H2 Nutrient inflows into the Dutch river delta

For the modelling of nutrients throughout the Dutch aquatic ecosystems. In order to model the flow of nutrients through water the pollution sources must first be established. N and P nutrients end up in the Dutch aquatic ecosystems through natural and anthropogenic processes. These pollution sources are modelled as inflows into either of the identified waterbodies (Fig. H1). First of all, nutrients are present in the water through natural processes of decomposition by microbial organisms. These processes occur in every body of water where biomass is present. Secondly, nutrients end up in water by means of anthropogenic pollution sources. These pollution sources originate either from the atmosphere, the soil, water or industry related activities including sewage treatment plants (Ministerie van Volksgezondheid, 2016).

P and N pollution of water originate from the same sources, with the exception of the atmosphere as phosphorus pollution through the air is negligible. Atmospheric N pollution of the water occurs through arial deposition of Nr. The inflow of Nr through arial deposition is modelled based on the surface area of each waterbody. Pollution through the soil occurs through the aquatic processes of leaching and run-off, which transport nitrogen and phosphorus from soil to water. Agricultural practices of manure and fertilizer application on culture grounds are a major source of aquatic nitrogen and phosphor pollution. Moreover, transboundary water which holds a concentration nitrogen and phosphorus is a significant contributor to the inflow of nitrogen and phosphorus into the Dutch river delta. Lastly, industries and sewage treatment facilitate a significant impulse of nitrogen and phosphorus to the Dutch river delta.

H3 Nutrient flows throughout the Dutch river delta

Nutrients flow throughout the Dutch river delta are modelled following the water flows depicted in Figure 4. The water which flows from one waterbody to another transports the nutrients. The amount of nutrients which is transported depends on the volume and the concentration of the water that is transported. The concentration is derived from the amount of nutrients which is present in a respective water body and the volume of the waterbody. The amount of nutrients which then flows to the next waterbody is determined based on this concentration and the spill-over volume of water into the next waterbody. Since the spill-over water is less than the total inflow of water in each waterbody due to evaporation, a portion of the nutrients does not flow on to the next waterbody. This implies that a part of the nutrients remains inside of the waterbody, independent of the flow of water. As factors such as sedimentation are not incorporated in the model, the remaining nutrients are deemed viable.

H4 Aquatic biomass accumulation and decomposition

Similar to soils, nutrients become available or unavailable in waterbodies through processes of biomass assimilation and decomposition. The growth of biomass is impacted by the availability of N and P nutrients, as these nutrients provide the fundamental building blocks for biomass accumulation (Chapin III et al., 2011). Additionally, the state of biomass in the waterbodies impacts the growth of biomass. The availability of nutrients impacts accumulation in two ways, namely through phosphorus availability and nutrient limitation.

Phosphorus availability impacts biodiversity in aquatic ecosystems as it is found to be the major driver for eutrophication in these areas. The assumption is made that all waterbodies in this research are in a state of phosphorus limitation, as suggested in literature (Djodjic et al., 2004) . When phosphorus is abundantly available in waterbodies under these circumstances, it stimulates the growth of phosphoric biomass. In phosphorus rich waterbodies, the phosphoric plants are thereby at a competitive advantage, resulting in a relatively faster growth of phosphoric plants compared to the other aquatic biomass. This dynamic is incorporated in the model with the use of the “Phosphorus impact variable”, which is determined separately for each waterbody. The variable represent the impact of phosphorus availability on phosphoric plant growth based on the “phosphorus impact on growth function”. The function is derived from KRW which presents favourable nutrient concentrations in water to preserve biodiversity (Ministerie van Volksgezondheid, 2016). Since the function is estimated based on a single presented norm, a correction factor is added to generate viable assimilation rates and test the impact of the function.

In addition to nutrient availability, nutrient limitation impacts the growth of biomass based on the relative availability of nutrients. As biomass requires both N and P nutrients for growth, limitation of either one reduces the amount of potential biomass growth. This dynamic is incorporated in the aquatic biodiversity sub-models with the "Aquatic N:P ratio function", which is based on the assumption that aquatic ecosystems are nitrogen limited. With this function the impact of nutrient limitation on growth is determined based on the N:P ratio. The function itself is derived from literature data on the impact of nutrient limitation on plant growth. It emulates the relation where low N:P ratios indicate low degrees of P limitation and thus high overall biomass growth. Conversely, high N:P ratios indicate high degrees of P limitation and thus low utilization of the biomass growth potential.

Appendix I - Biodiversity

For Dutch policy on nature preservation to be successful, it must protect the biodiversity and habitats in the Natura 2000 areas (European Commission, 2018, 2021). Currently, the state of the Nature 2000 areas are in deemed insufficient, and changes are required to restore the state of these areas and its biodiversity to a favourable condition (Adviescollege Stikstofproblematiek, 2020). To test the success of policy options within the renewed policy framework, sub-models are setup to indicate the state and performance of biodiversity in aquatic and terrestrial Nature 2000 areas. The state of biodiversity is measured based on the share of desirable plants in the biomass of Natura 2000 areas. The measure is a simplistic approach to the measure of biota diversity as suggested by Purvis (2000).

The state of biodiversity is impressionistically modelled to generate plausible model outcomes. This means that the structure and values chosen for variables underlying growth, decay and biomass potential are estimated and chosen to produce impressionistic model behaviour. The values are therefore not based in scientific literature, but adjusted to meet expected model behaviour. This impressionistic approach is deemed necessary considering a lack of readily available research on the impact of nutrient pollution on biodiversity required for a detailed analysis of biodiversity and its relations to nutrient availability and soil and water dynamics. Moreover, this approach is deemed sufficient considering the goal of this research, which is to provide a proof of principle, and a basis for further research.

I1Terrestrial

For the determination of the state of biodiversity in terrestrial ecosystems, a distinction is made between nitrophilic biomass and other terrestrial biomass. This distinction is made based on the assumption that all terrestrial ecosystems are nitrogen limited (P. Vitousek & Field, 2001). These ecosystems are thereby vulnerable to nitrogen pollution. An increase of nitrogen pollution in these ecosystems benefits the growth of nitrophilic plants over other terrestrial plants, which is harmful to the state of biodiversity (Chapin III et al., 2000; Lambers et al., 2011). Following this logic, the state of terrestrial biodiversity is determined as the relative share of other terrestrial biomass compared to the total amount of terrestrial biomass, which includes the “other terrestrial biomass” and the “nitrophilic biomass”. The measure for the state of terrestrial biomass thereby takes a value between 0 and 1, where a value closer to 1 indicates a more favourable state of biodiversity.

The state of terrestrial biodiversity emerges from processes of biomass growth and biomass decay. These processes alter the composition of terrestrial biomass, from which the state of biodiversity emerges. These processes work similarly for the nitrophilic biomass stock as for the other terrestrial biomass stock.

The growth of biomass is determined based the terrestrial biomass growth factor, the ratio of N:P, the impact of nitrogen availability, and the relative potential for biomass growth. As explained in (Appendix F), the ratio of N:P is used to determine the limitation of the total biomass growth through the "terrestrial N:P ratio function". Additionally, the "nitrogen impact" variable is used to determine the impact of nitrogen on the growth of nitrophilic biomass and other terrestrial biomass (Appendix F).

The relative potential for biomass growth limits the amount of biomass that can grow in the Nature 2000 areas based on the maximum amount of biomass that can be held in an area and the current amount of biomass which is present. The relative potential for biomass growth is thereby critical for ensuring biomass cannot accumulate limitlessly in the model. the initial state of biodiversity is chosen to be non-favourable, meaning that half of the of biomass is nitrophilic in its initial state. Moreover, the assumption is made that the majority of biomass potential in terrestrial ecosystems is already in use and the relative potential for further growth of either form of biomass is thereby low. Lastly, the decay of both nitrophilic and other terrestrial biomass is simply modelled based on an estimated lifetime for the types of biomass.

I2 Aquatic

The state of biodiversity in aquatic ecosystems is determined based on the same model structure as used for the terrestrial biodiversity. The state of biodiversity in aquatic ecosystems is determined based on the relative share of "other aquatic biomass" compared to the total amount of aquatic biomass, encompassing "other aquatic biomass" and "phosphoric biomass". Based on the assumption that aquatic ecosystems are phosphor limited (Djodjic et al., 2004), the distinction between "phosphoric biomass" and "other aquatic biomass" is made. Similar to the terrestrial measure for biodiversity, the state of aquatic biomass takes a value between 0 and 1, where a value closer to 1 indicates a more favourable state of biodiversity. A relative growth of "phosphoric biomass" compared to "other aquatic biomass" thereby reduces the overall state of aquatic biodiversity.

The state of aquatic biodiversity emerges from processes of biomass growth and biomass decay. The decay of biomass is, similar to terrestrial biomass, modelled simply based on an estimated lifetime for each type of biomass. The growth of biomass is however more complex as it is influenced by a multitude of factors, namely; aquatic biomass growth factor, the ratio of N:P, the availability of phosphorus and the relative potential for biomass growth (Appendix F). Similar to the terrestrial ecosystem, the ratio of N:P is used to determine the impact of P limitation on the growth of biomass and is modelled through the “aquatic N:P ratio function”. This function describes the relation between P limitation and biomass growth, as explained in Appendix F.

To model the impact of phosphor availability on the state of biodiversity, the variable “phosphor impact” is incorporated in the model. The “phosphor impact” variable is used to determine the distribution of biomass growth over both forms of biomass, based on the availability of phosphor in the water. A higher availability of phosphor results in relatively more “phosphoric biomass” growth and less “other aquatic biomass” growth.

Lastly, the relative potential for aquatic biomass growth is used to limit the amount of biomass that can accumulate in a waterbody. This potential is based on the estimated total volume of biomass that can grow in each waterbody and the amount of biomass which is present at any time. The relative potential for aquatic biomass growth is thereby critical to limit the amount of biomass accumulation which can occur. The initial state of aquatic biodiversity and the utilization of biomass for each waterbody is determined separately for each waterbody based on their expected state of biodiversity and utilization of biomass potential. These values are estimated based on reports of state of biodiversity in these areas and the degree of nutrient pollution in these areas (Willems et al., 2008).

Appendix J – Traffic

Traffic is a major source of NO_x emissions in the Netherlands. NO_x emissions originate from the combustion processes which occur in fossil fuel cars. For the modelling of NO_x emissions from traffic a distinction is therefore made between fossil fuel based cars (e.g. diesel and gas), and electric cars which do not emit NO_x. To determine the amount of NO_x emissions which occur from the fleet, the composition of the fleet has to be modelled over time. To this end for each type of car (e.g. diesel gas or electric) a stock is setup which changes based on an inflow and outflow of cars. The outflow of cars is modelled based on the amortization of cars which is based on the average car lifetime. Conversely, the total inflow of cars is based on the demand for new cars, and distributed per car type is based on a choice factor for each type of car. This choice factor is influenced by the Dutch policies which aim to increase the electric car use, and thereby shape the changes in fleet composition.

The amount of NO_x emissions that occur from the fleet are then dependent on the distance driven, where logically more distance travelled results in more NO_x emissions. Additionally, the speed of travel impacts the amount of NO_x emissions, where higher speeds result in more NO_x emissions per kilometre. The amount of NO_x emissions is eventually determined based on the total driving kilometre for both gas cars and diesel cars, and the amount of NO_x emissions per km which occurs at a certain speed limit for diesel and gas cars.

Appendix K – Livestock and Manure

The flows of N and P throughout the livestock sector are modelled based on data presented in the nitrogen and phosphor flow diagram (CBS, 2020b). These flows are setup separately for N and P. The inflow of nutrients in this sector originate from the feed that is fed to the cattle and the fertilizers that are put into the soils for the cultivation of crops. The amount of feed that flows into the livestock are based on the demand of livestock. Feed comes in two forms; rough feed and power feed. Rough feed originates from the crops that are cultivated in the culture grounds. Additionally, Power feed is fed to the cattle to satisfy their food demand. The assumption is made that the demand for rough feed and power feed remains proportionate to the amount of cattle. In case the plant produce from culture grounds is no longer able to meet the rough feed demand of livestock, the assumption is made that this rough feed will be imported. The demand for food is deduced from the total amount of cattle units and the total amount of feed that was fed to the livestock.

The amount of manure that is present in the sector is dependent on the amount of nutrients present in livestock. The nutrients in livestock are either transformed into animal produce or manure. The share of manure and animal produce that is produced from livestock is assumed to remain constant. Manure is then used in large to meet the manure demand for the cultivation of crops in culture grounds. The assumption is made that the amount of manure used by farmers will always be equal to the norm set by governments, as it can be expected that farmers aim to maximize their crop yield. In case the manure produced by livestock does not meet the demand for manure application, manure has to be imported. Otherwise, the excess of manure is exported.

Appendix L – Policies

To analyse the impact of government policy these policies have to be incorporated in the model. As discussed in Paragraph 3.1, the following policies are considered in this research. For each policy measure the link to the model and the determined parameter values are expanded upon below.

L1 Speed limit

The speed limit is used by the government as a policy instrument to reduce NO_x emission from traffic. A lower speed limit results in less NO_x emissions per kilometre. A restriction of the speed limit thus lowers the amount of NO_x emissions which occur from traffic. The policy is parameterized at a 100 or 130 km/h, with 130 km/h being the regular speed limit in the Netherlands and a 100 km/h the one which was opted for in 2019 to mitigate the effects of the nitrogen crisis. Based on data from CE Delft (den Boer & Vermeulen, 2004) the amount of NO_x emissions per km for each speed limit is determined. With the policy on speed limit as input, the model selects the corresponding NO_x emission variable and from there calculates the total NO_x emissions which occur from traffic. The cost related to speed reduction are implicit by means of a reduction of economic activity, and does not require a direct investment to implement. The estimation of such indirect costs are ambiguous which is why the assumption is made that the policy cost related to policy on speed limit are zero.

L2 Livestock reduction

Livestock is the source of the majority of domestic NH_3 emissions through its production of manure (CBS, 2020b). By reducing the livestock, the government can significantly reduce the amount of nutrient intake by cattle and consequently the amount of manure that is produced. This policy is implemented in the model through the use of the variable “effective livestock cattle”, which uses the policy variable “Policy livestock reduction” as input. Based on the input value the “effective livestock cattle” value is either altered based on the policy inputs. The policy variable is parameterized for 1 and 0.5, which corresponds to 0% and 50% reduction of livestock. The cost for livestock reduction is estimated based on reports in the NRC which indicate a total cost of 10 to 17 billion euro’s for the reduction of livestock (Kuiper & Rutten, 2021).

L3 Manure application

The policy for manure application is aimed at reducing nitrogen emissions from culture grounds by reducing the amount of manure application. This policy is implemented by reducing the norm for manure application which consequently results in a smaller inflow of manure to the culture grounds. This policy is parameterized at 1 and 0.5, indicating a 0% and 50% reduction of manure application.

The cost this policy are estimated relative to the cost for the reduction of livestock. The cost for reducing the amount of manure that farmers can apply directly impacts the amount the crop yield of farmers and hence their profitability. Compensation for such policies are deemed necessary and assumed to halve of that required for the livestock reduction. The assumption is made based on the size of agriculture and livestock sector, which is comparable in terms of added value (Adviescollege Stikstofproblematiek, 2020, p.31). Additionally, the assumption is based on the expectations that agriculture farmers do not have to be bought out of their lands, and they still have options to ensure crop yield through fertilizer application and the remaining 50% of manure application.

L4 Agriculture transition

A clear policy direction presented and enacted by government is the transition to a low-emission agriculture sector. This policy pathway is aimed at improving livestock stables and manure storage systems to reduce NH_3 emissions. Yearly investments to this cause have been announced to add up to 170 million euros per year. The policy is integrated in the model through the "livestock farms transition" sub-model. The sub-model determines the yearly transition of farms and their consequent potential for emission reduction. The policy parameterized at 170 million and 340 million per year to represent the already enacted base rate of 170 million (Global Agricultural Information Network, 2020) and analyse the impact of speeding up the process of agricultural transition.

L5 Turfing

Turfing removes the top layer of the soil and consequently the nutrient and plants in it. Turfing therefore creates an outflow of N and P from Natura 2000 soil and biomass stocks. The effect of turfing on biomass stocks is simply the equal to the amount of biomass which is present in a certain area. A complication which has to be accounted for is the non-uniform impact of turfing on biomass removal over an area of nature. Namely, turfing removes all the plants in one area whilst leaving another completely untouched. This dynamic has to be compensated for to ensure valid modelling of the effectiveness of turfing on the removal of biomass. Accordingly, this is done by keeping a record on the share of areas that have already been turfed, thereby knowing share of non-turfed area and the amount of biomass present in it.

The effect of turfing for the removal of nutrient in soils is dependent on a multitude of factors. Firstly, turfing removes nutrient dense top layer of the soil, and thereby a substantial share of the nutrient content of the soil. The effectiveness of turfing for nutrient removal in an area of soil is thereby not 100%. This is accounted for by the “turfing effectiveness for nutrient removal” variable. Moreover, a similar dynamic is present for the turfing of soils as for the turfing of biomass relating to the nutrient content of soil. To ensure proper modelling of the effectiveness of turfing for the removal of nutrients from the soil a record has to be held on the share of areas that have already been. Additionally, a restriction on the amount of turfing is deemed necessary due to its drastic implication on nature, which is not likely not to be allowed in most areas of the Natura 2000 network. Lastly, the amount of turfing that occurs on a yearly basis is dependent on the investments made for turfing operations, the cost for turfing per area and the area that is allowed to be turfed. Similar to the other policy options, the available budget for turfing practices is parameterized at 200 million per year.

L6 Landscaping

Landscaping refers to the practice of removing plants from an ecosystem with the aim of improving its biodiversity. Landscaping is thereby aimed at plant species that are harmful to the ecosystems biodiversity. Based on the assumption that terrestrial ecosystems are nitrogen limited, the unwanted plant species refer to nitrophilic plants such as nettles, wisteria or blackberry bushes. Landscaping hereby does not impact the nutrients in soil but only in biomass stocks. The removal of such plants must be done by hand. The outflow of nutrient from biomass is there for determined based on an estimation for the cost of “gardeners”, the area they can cover per year and the budget available for landscaping practices.

L7 Dredging

Dredging refers to the practice of removing nutrient rich slip and plants from ditches and trenches adjacent to culture grounds. Such waterbodies are directly impacted by the nutrient leaching and run-off from culture grounds, which eventually flow downstream into the regional and larger waterbodies. The amount of nutrient that are removed through dredging practices is thus dependent on the amount of plants and slip that is removed from the ditches. Firstly, the amount of nutrient removal depends on the available funds for dredging practices which are estimated at 200 million per year, similar to the turfing and landscaping measures. From estimates on the km cost of dredging, the yearly coverage of dredging practices can be determined. It is assumed that locks that are dredged are completely removed from plants. Contrarily, the share of nutrient that are removed by means of slip is estimated at around 80%. The assumption is based on the notion that the top layers of soils hold the highest concentration of nutrients, thereby making dredging an effective measure for the removal of nutrients from the soils of trenches.

Appendix M - Nitrogen

Nitrogen can take on a variety of compositions throughout its cycle (Berhe et al., 2010). The nitrogen compositions most relevant to the issue of nutrient pollution are ammonia (NH_3) and NO_x (Adviescollege Stikstofproblematiek, 2020), the nitrogen content of which is referred to as reactive nitrogen (Nr). These nitrogen compositions are key for plant growth, which makes them critical for high crop yield in agriculture , but often degenerative to biodiversity. Both NH_3 and NO_x can occur in a gaseous state and can thereby be transported through air. Nr in the shape of NO_x originates either from combustion processes, or denitrification processes occurring in water or soils. Nr in the form of NH_3 occurs in the atmosphere through the process of volatilization, which originates predominantly through agricultural practices.

There are three major pathways for Nr to reach the soil, through; arial deposition, microbial N2 fixation or application of manure and fertilizer. When Nr is absorbed into the soil, the majority of NO_x and NH_3 assimilate into biomass or leaches to ground and surface water in the form of nitrate (NO_3^-). Additionally, NO_x and NH_3 can return back to the atmosphere through volatilization or denitrification. Volatilization is the process through which NH_3 escapes the ground after the application of manure or fertilizer. Denitrification is a chemical processes which returns NO_x in the soil, back to the atmosphere in the form of N2. Denitrification is enhanced in case of high Nr availability, as is the case when manure or fertilizer are applied.

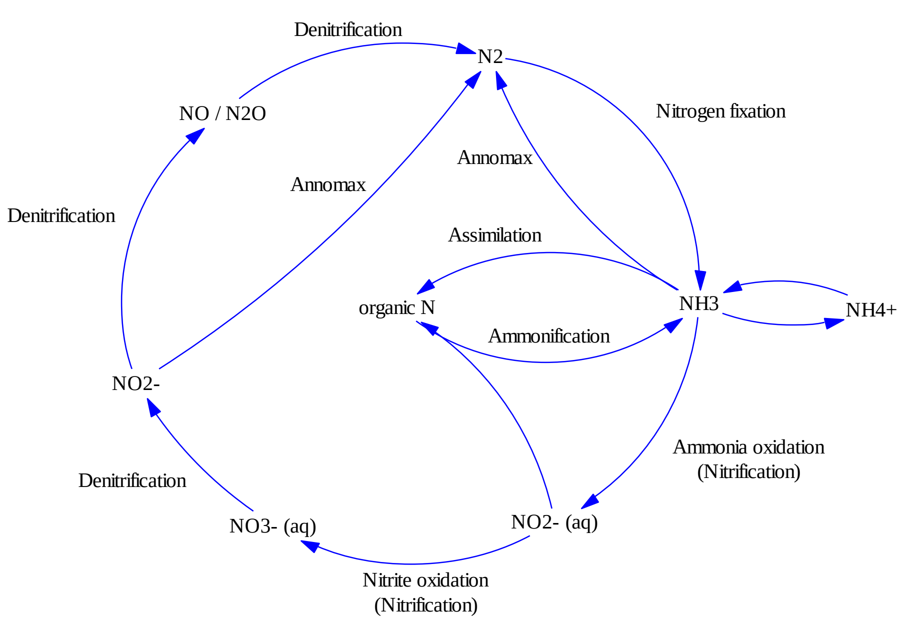


Figure M: The nitrogen cycle derived from Bernhard (2010)

Nr reaches water through arial deposition or soil leaching in the form of nitrate (NO_3^-). Soil leaching occurs when Nr is not assimilated, denitrified or volatilized. When present in water bodies, Nr can result in; assimilation through aquatic biomass growth, microbial denitrification or sedimentation. When high concentrations of Nr are present in waterbodies, eutrophication occurs through rapid aquatic biomass growth in the form of algae blooms. Eutrophication occurs when an algae bloom results in a state of oxygen scarcity in the water, thereby resulting in a loss of biodiversity. When Nr in water bodies does not assimilate it is either sedimented to the soil, or denitrifies to the atmosphere. Additionally, Nr which is present in waterbodies can leave the domestic system through outflow into the ocean.

Appendix N - Phosphorus

Phosphorus can take on a variety of forms throughout its cycle (Berhe et al., 2010). The most relevant form of P is when it is in its dissolved form, also referred to as “soil solution P”. The majority of soil solution P becomes available through two predominant pathways, namely; manure and fertilizer application, or decomposition of organic material. Another pollution pathway of phosphorus occurs in the form of phosphorus rich dust, which allows phosphorus to be transported over short distances through air. This form of transportation only occurs over short distances and is highly dependent on meteorological factors and agricultural practices. In literature the arial pollution pathway of phosphorus is often deemed as insignificant (Berhe et al., 2010). For these reasons the arial pathway of phosphorus is not considered in this study.

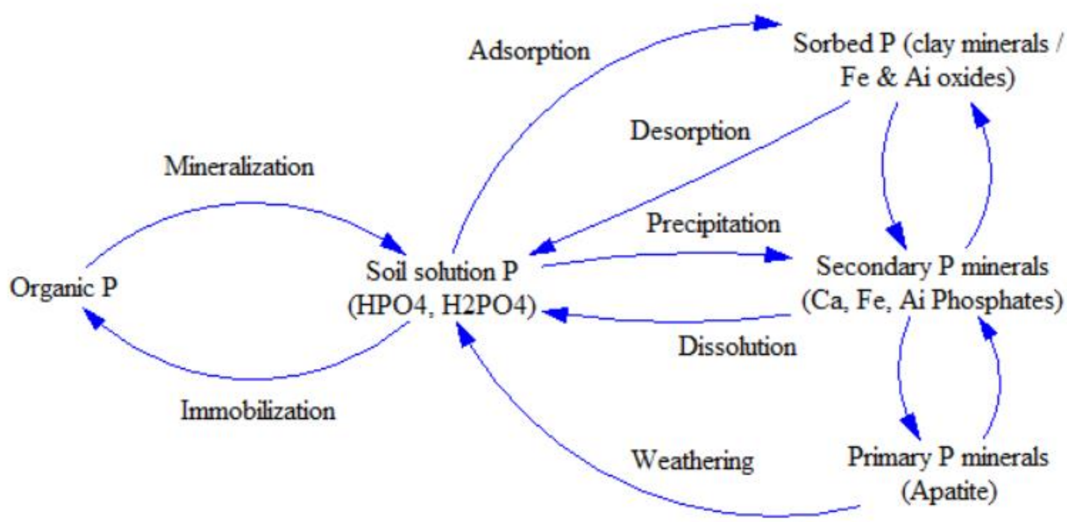


Figure 3: the phosphorus cycle as derived from Tian (2021)

When soil phosphorus is in its dissolved state it is available for plant uptake or assimilation into organic matter. When phosphorus is mineralized or assimilated into organic matter or minerals, it becomes highly immobile. Only through slow processes of desorption, mineralization or weathering can P from these stocks become available again. If soil solution P is not immobilized, it can leach to ground and surface water. When phosphorus is transported to water bodies, it can result in rapid growth of aquatic biomass. When aquatic biomass growth exceeds a certain threshold, its growth can result in eutrophication and a loss of biodiversity in these waterbodies. A study by identified P pollution as the main driver for eutrophication in most waterbodies (Foy, 2015) .

Appendix O – Model uncertainties

The table below lists the uncertainties that are incorporated in the model. The table indicates source and the minimum and maximum value for each of the variables.

Table O: Overview of the model's uncertainties variable

Uncertainty variable	Units	Min	Max	Source
Initial concentration NO_x in atmosphere in kg	kg/(km*km*km)	12.6	15.4	RIVM (2020c)
Initial concentration NH_3 in atmosphere in kg	kg/(km*km*km)	5.76	7.04	RIVM (2020a)
Foreign NH_3 inflow	ktn/Year	30.16	44.64	TNO (2019)
Foreign NO_x inflow	ktn/Year	180.7	271.08	TNO (2019)
NO_x deposition factor	Dmnl	0.193	0.266	Derived from TNO and RIVM data
NH_3 deposition factor	Dmnl	0.421	0.515	Derived from TNO and RIVM data
Initial N concentration soil	ktn/(km*km)	0.1	0.15	Goed Bodembeheer (2021)
Residence time N NCG	Year	32	48	Assumption
Denitrification rate NCG soils	kg/(km*km)/Year	90	110	Barton (1999)
Initial CG volatilization factor	Dmnl	0.79	0.97	CBS (2020b)
Initial crop yield efficiency N	Dmnl	0.54	0.66	CBS (2020b)
CG denitrification rate	Dmnl	0.11	0.134	CBS (2020b)
CG N leaching to surface water %	Dmnl	0.08	0.12	Estimated based on Oenema (2005)
CG residence time N	Year	12.8	19.2	Assumption
CG N background leaching factor	Year	176	264	Estimated based on Oenema (2005)
CG N deep seapage factor	Year	8	12	Assumption
Initial P concentration soil	kg/kt	12	18	Goed Bodembeheer (2021)
Residence time P NCG	Year	32	48	Assumption
Initial crop yield efficiency P	Dmnl	0.608	0.912	CBS (2020b)
CG P leaching to surface water %	Dmnl	0.12	0.18	Estimated based on Chardon (2002)
CG residence time P	Year	11.2	16.8	Estimated based on Chardon (2002)
CG P background leaching factor	Year	200	300	Estimated based on Chardon (2002)
CG P deep seapage factor	Year	40	60	Estimated based on Chardon (2002)
Uncertainty crop yield efficiency growth	Dmnl	0.9	1	Assumption
Agricultural waterflow	(km*km*km)/Year	10.8	13.2	RIVM (2003)
Spill-over factor agri to regional	Dmnl	0.82	1.01	RIVM (2003)
Waterflow rainfall to regional water	(km*km*km)/Year	13.5	16.5	RIVM (2003)
Spill-over factor regional to rivers	Dmnl	0.83	1.01	RIVM (2003)

Foreign water inflow	(km*km*km)/Year	70.2	85.8	RIVM (2003)
Spill-over factor large rivers to ocean	Dmnl	0.765	0.935	RIVM (2003)
Initial N concentration agri water	ktn/(km*km*km)	3.051	3.729	RIVM (2020b)
Initial N concentration regional water	ktn/(km*km*km)	2.034	2.486	RIVM (2020b)
Initial N concentration large rivers	ktn/(km*km*km)	2.44	2.98	RIVM (2020b)
N concentration foreign water	ktn/(km*km*km)	2.28	3.42	Derived from RIVM (2003, 2020b)
Initial P concentration agri water	ktp/(km*km*km)	0.36	0.44	RIVM (2020b)
Initial P concentration regional water	ktp/(km*km*km)	0.252	0.308	RIVM (2020b)
Initial P concentration large rivers	ktp/(km*km*km)	0.135	0.165	RIVM (2020b)
P concentration foreign water	ktp/(km*km*km)	0.088	0.132	Derived from RIVM (2003, 2020b)
Direct N loading from agriculture	Ktn/Year	2.34	2.86	RIVM (2020b)
Industry N to regional water	Ktn/Year	1.08	1.32	RIVM (2020b)
STP N to regional water	Ktn/Year	6.3	7.7	RIVM (2020b)
Industry N to large rivers	Ktn/Year	1.08	1.32	RIVM (2020b)
STP N to large rivers	Ktn/Year	6.3	7.7	RIVM (2020b)
Direct P loading from agriculture	Ktp/Year	0.36	0.44	RIVM (2020b)
STP P to regional water	Ktp/Year	0.99	1.21	RIVM (2020b)
Industry P to regional water	Ktp/Year	0.09	0.11	RIVM (2020b)
STP P to large rivers	Ktp/Year	0.99	1.21	RIVM (2020b)
Industry P to large rivers	Ktp/Year	0.09	0.11	RIVM (2020b)
Initial utilization of terrestrial biomass capacity	Dmnl	0.7	0.9	Assumption
Initial share of nitrophilic biomass	Dmnl	0.4	0.6	Assumption
Other terrestrial biomass lifetime	Year	35	45	Assumption
Nitrophilic biomass lifetime	Year	22.5	27.5	Assumption
Other aquatic biomass lifetime	Year	7	9	Assumption
Phosphoric biomass lifetime	Year	4.5	5.5	Assumption
Initial utilization of aquatic biomass capacity AW	Dmnl	0.7	0.9	Assumption
Initial share of phosphoric biomass AW	Dmnl	0.64	0.96	Assumption
Initial utilization of aquatic biomass capacity RW	Dmnl	0.56	0.84	Assumption
Initial share of phosphoric biomass RW	Dmnl	0.48	0.72	Assumption
Initial utilization of aquatic biomass capacity LR	Dmnl	0.16	0.24	Assumption
Initial share of phosphoric biomass LR	Dmnl	0.24	0.36	Assumption
Terrestrial N:P function correction factor	Dmnl	0.8	1.2	Assumption
Aquatic N:P function correction factor	Dmnl	0.77	0.93	Assumption
N to P assimilation ratio	ktn/ktp	11.25	13.75	Assumption
Initial N:P availability ratio	ktn/ktp	13.5	16.5	Assumption

Nitrogen impact correction factor	Dmnl	0.9	1.1	Assumption
Phosphorus impact correction factor	Dmnl	0.9	1	Assumption
Terrestrial biomass to nitrogen conversion factor	ktn/kt	0.0008	0.0012	Assumption
Aquatic biomass to phosphorus conversion factor	ktp/kt	0.00008	0.00012	Assumption
Terrestrial biomass growth factor	1/Year	0.36	0.44	Assumption
Aquatic biomass growth factor	1/Year	1.8	2.2	Assumption
Cattle unit cow	VEE/Animal	0.9	1.1	NVWA (2021)
Cattle unit pig	VEE/Animal	0.18	0.22	NVWA (2021)
Cattle unit poutly	VEE/Animal	0.00603	0.00737	NVWA (2021)
Powerfeed P per cattle unit	ktp/VEE/Year	8.7534e-6	1.07e-5	Derived from NVWA (2021) and CBS (2020b)
Powerfeed N per cattle unit	ktn/VEE/Year	5.18e-5	6.32e-5	Derived from NVWA (2021) and CBS (2020b)
Roughfeed P per cattle unit	ktn/VEE/Year	4.81e-6	5.88e-6	Derived from NVWA (2021) and CBS (2020b)
Roughfeed N per cattle unit	ktn/VEE/Year	3.587e-5	4.385e-5	Derived from NVWA (2021) and CBS (2020b)
Initial N in livestock	Ktn	640	782	CBS (2020b)
Initial Manure N	Ktn	453	554.4	CBS (2020b)
Initial plant produce N	ktn	315	385	CBS (2020b)
Manure volatilization factor	1/Year	0.1018	0.1244	CBS (2020b)
Livestock to manure N factor	1/Year	0.639	0.781	CBS (2020b)
Initial P in livestock	Ktp	99	121	CBS (2020b)
Initial Manure P	Ktp	63.9	78.1	CBS (2020b)
Initial plant produce P	Ktp	43.2	52.8	CBS (2020b)
Livestock to manure P factor	1/Year	0.58	0.71	CBS (2020b)
Average driving distance diesel cars	km/(Year*Car)	20293.2	24802.8	CBS (2020a)
Average driving distance gas cars	km/(Year*Car)	9787.5	11962.5	CBS (2020a)
Growth factor car demand	1/Year	1.0035	1.0045	Assumption
Average car lifetime	Year	18	22	Assumption
Initial amortization factor	1/Year	1.1	1.5	Assumption
Uncertainty NO_x shipping reduction trend (post 2020)	Dmnl	1	1.2	Adviescollege Stikstofproblematiek (2020)
Uncertainty NO_x industry reduction trend (post 2020)	Dmnl	0.9	1.1	Adviescollege Stikstofproblematiek (2020)
Uncertainty NO_x consumers, services, government and construction reduction trend (post 2020)	Dmnl	1	1.15	Adviescollege Stikstofproblematiek (2020)
Uncertainty NO_x agriculture and livestock trend (post 2020)	Dmnl	1	1.2	Adviescollege Stikstofproblematiek (2020)
Uncertainty NH_3 reduction trend (post 2020)	Dmnl	1	1.2	Adviescollege Stikstofproblematiek (2020)
Dredging effectiveness for nutrient removal	Dmnl	0.64	0.96	Assumption

Cost per km of dredging	mln/km	0.001	0.003	Offerte Adviseur (2021)
Maximum turfing norm	Dmnl	0.4	0.6	Assumption
Turfing cost per km2	mln/(km*km)	0.8	1.2	Assumption
Turfing effectiveness for nutrient removal	Dmnl	0.56	0.84	Assumption
Landscaping effectiveness	Dmnl	0.36	0.65	Assumption
Area per gardener	(km*km)/ (gardener * Year)	0.288	0.432	Assumption
Cost per gardener	mln/gardener/Year	0.08	0.12	Assumption
Effectiveness Dutch electric car policy	Dmnl	0.5	1	Assumption
Initial share of transitioned livestock farms	Dmnl	0.105	0.195	Assumption
Average farm transition cost	mln/Farms	0.8	1.2	Assumption
Potential for manure volatilization reduction	Dmnl	0.5	0.8	Expert interview : Jan v. Mourick
Livestock reduction cost scenario	Dmnl	1	1.9	Kuiper (2021)

Appendix P – KDW exceedance map of the Netherlands

Figure P: Map of KDW exceedance of the Netherlands in 2030 (RIVM, 2020b)