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- a Cost Benefit Analysis applied on the fish farming industry on Åland

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Abstract

One of the present-days largest challenges are the increased climate change and the scarcity of non-renewable resources. From both global, national and local levels comes exhortations concerning lowering energy use from fossil fuels and streamlining resources more efficiently. An important solution to attack this problem is to increase the use of biofuels that leave a lower carbon footprint and that is not threatened by scarcity (if consumed in a responsible way). Developing more advanced biofuels, biofuels that are made from residues or waste, could create multiple environmental advantages and make exploitation of resources more effective and sustainable.

Åland, the self-governing island between Sweden and Finland, is a small economy with an essential fish industry. Every year this island, with less than 30 000 inhabitants, produce fish for more than 300 000 human's consumption in their fish farms every year. With that production comes a lot of waste that currently is mostly sold to animal farms in mainland Finland.

This thesis will investigate whether Åland have a better use for the waste residual from their fish farming. The intent is to compare the current applications for fish waste to a biogas scenario where the fish waste is used for anaerobic digestion to make biogas upgraded to vehicle gas. The first objective is to identify theoretical impacts from the biogas scenario on society's utility in relation to status quo. Both internal and external effects are recognized. The second objective is to value this scenario using a Cost Benefit Analysis where the impacts are monetized and evaluated over time to find whether it is socio economically profitable to applicate biogas production of fish waste.

The result showed a negative Net Present Value for biogas production from fish waste of EUR 2 552 853 over a studied 40-year period. In the sensitivity analysis, the robustness of this result was tested and showed that the result stayed below zero regardless of whether the time frame was increased or whether the benefit for reducing environmental degradation was increased with time. The sensitivity analysis did however find that the result is significantly sensitive to different assumptions regarding the estimated benefit of reduced greenhouse emissions and to the production costs of biogas. If the total production of biogas could increase the production cost per kWh (kilowatt-hours) would reduce to the extent where biogas production from fish waste would break even with the status quo.

Further research is recommended to include more impacts that were recognized but not monetized within this study and to develop customized economic instruments that would adjust for the external effects of producing biogas from waste products such as fish waste.

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1. Chapter One - Introduction

1.1. Introduction and Problem Statement

In a contemporary world where climate change and resource scarcity have emerged into a growing problem on the world leaders' agenda, the importance of sustainable development and resource efficiency is identified as crucial fundamentals towards a better future. One effective way of achieving this is to increase the share of energy coming from biofuels and to recirculate resources making more value from waste (UNEP, 2010).

In Finland, there is a goal of increasing the share of biofuels in the transport sector to 30% before 2030 (Huttunen, 2017). Sweden has set the bar even higher and work towards the government set goal of a fossil free transport sector by 2030 (Wallmark et al., 2016) Replacing fossil alternatives with biogas in the transport sector have clear environmental advantages on emissions, both greenhouse gas emissions that have negative effects globally, and emission of particles that have negative effects locally (Brännlund et al, 2010).

On the global market, 95 percent of the biofuels come from crops that could have been used for food production (food-based biofuels) (European Commission, 2014). The European commission (2014) has acted to decrease the usage of the food-based biofuels and invest in advanced biofuels instead by withdrawing support for food-based biofuels. Advanced biofuels are defined by the EU as biofuels made from residues, waste or non-food-based cellulose and lignin (ibid). To increase the share of biofuels in the transport sector is a difficult task while at the same time incorporating the expansion of advanced biofuels. In this context, it is of great relevance to develop and research the economic potential for new possible bases of biofuels.

For Åland, more local level sustainable goals have been developed and the Government of Åland have established a climate strategy initiated in a report 2007 where they state that their vision is that 100% of Åland's energy production would be carbon dioxide free (Wiklund, 2007). A requirement for achieving this is, according to the report, to increase energy coming from biofuels.

Farmed fish is Ålands undoubtedly largest export commodity. Each year, Åland fish farms sell more than 13 000 tons of fish which is equivalent to 300 000 humans consumption (Åland's Seafood Growers Association, 2017). The fish production entails a lot of bi-products in the form of waste which measures just above 2 500 ton per year. Today most of this waste is either composted or sold as feed for animal farming (ibid).

Fish waste is a very rich waste product containing both fats and protein that is well adapted for producing biogas vehicle fuel through anaerobic digestion (Tufvesson and Lantz, 2012). If the produced biogas replaces fossil fuels in the transport sector this could contribute to the goal of replacing fossil fuels with biogas and to lower emissions of greenhouse gases and other emissions (ibid). Fish waste is highly ranked by the European Maritime and Fisheries Fund (EMFF) as an advanced biofuel since the opportunity cost is low for alternative use compared to food-based biofuels (Wiklund, 2007).

The rest product from anaerobic digestion contains a larger proportion of ammonium nitrogen which makes it an efficient fertilizer that can replace mineral fertilizer in agriculture. Since the bio fertilizer contain more efficient nitrogen the total leakage of nitrogen from agriculture to the sea can be reduced. This has an environmental value since nitrogen leakage can cause eutrophication (Brännlund et al, 2010). Producing biofuel from fish waste could hence lead to increased usage of non-food-based renewable resource, bettering the quality of the sea and improving the air.

The purpose of this study is to evaluate whether biogas production from fish waste is profitable on Åland, using a socioeconomic perspective. The possibility to use all fish waste from fish farms on Åland for biogas, first letting it undergo anaerobic digestion and later have it upgraded to vehicle fuel, will be evaluated compared to business as usual, i.e. current practices regarding use of the fish waste. The method for appraisal is a Cost Benefit Analysis where costs and benefits are identified and calculated. Both marketable impacts (transport costs, production costs and revenue from sales) and non-marketable impacts (reduced pollution of greenhouse gases, particles and nitrogen) will be considered. The net benefits are calculated for the society of Åland to evaluate the research question; *Is biogas production a socioeconomically profitable solution for reutilizing fish waste on Åland?*

1.2. Social and academic relevance

This research will be useful in decision making for whether Åland would be better off by utilizing their fish waste as biogas. The magnitude of the share of farmed fish production makes this analysis important from an infrastructural viewpoint. Above that it could be used in evaluations for other societies' reutilizing of fish waste or other aquatic substrates. Methodologically the thesis can contribute by highlighting the sensitivities of different economic evaluation methods and choice of discount rate in a real case study.

It should be noted that this thesis does not provide a full analysis of the socioeconomic benefits and costs from the scenarios, but it highlights a range of impacts, including environmental effects, and their economic values. The results from the study should therefore be seen as rough estimates of reality, rather than absolute truths.

1.3. Disposition of the thesis

The thesis is divided into nine chapters. Chapter two is meant to give insights of the cornerstones of the study, meaning information about Åland and its sustainable goals, fish farming and biofuels. Chapter three covers the earlier studies of the subjects. Chapter four covers the conceptual framework from where the study departs. The method for appraisal, CBA, is covered in chapter five with a specific focus on environmental CBA. Chapter six explains the contrasting scenarios, beginning with a review of status quo, the reference scenario. It continues with a description of the biogas scenario and a summary of the identified impacts. These impacts are later deeper explained one by one in chapter seven where the impacts quantification and valuation is stated and motivated. Chapter eight covers the result containing annual value and Net Present Value for the studied time frame. The chapter ends with a sensitivity analysis where the robustness of the result is tested by differentiating selected assumptions and values. Chapter nine is the final chapter where the research question is answered and where a discussion of the results is made, both regarding its reasonableness and its application possibilities. Finally, some comments on future studies are presented.

2. Chapter Two – Background

2.1. Case study Åland

Åland, a group of islands located in the Sea of Åland in-between the Baltic sea in the south and the Bothnian Sea in the north, is an autonomous and monolingual Swedish region of Finland (Parliament of Åland, 2017). Through its autonomy, which is ruled under Autonomy Act of Åland, the region have a special position in Finland. Åland have their own government, *Landskapsregeringen*, that exercises management in all areas under the Autonomy act, and their own parliament, *Lagtinget*, that have the right to legislate on their internal affairs and decide on the budget for the Åland Islands. Åland is demilitarized and neutralized implying that no military can remain on Åland and the islands may not be consolidated. Åland is a member of the EU and, like Finland, they use the currency Euro (EUR) (ibid).

The landscape consists of 6,700 islands and islets, even though 90% of the around 29 000 inhabitants live on the largest island, referred to as Mainland Åland (The Official Åland Website, 2017). The capital of Åland is Mariehamn, with its population of around 11 000 inhabitants, makes it the largest city on the islands. Among the areas that are defended by the autonomy act is nature and environmental conservation which implies that regarding environmental policy Åland acts as an independent state (Parliament of Åland, 2017).

2.1.1. Fish farming on Åland

For Åland, which consists of 90% water and 10% island landscape, fishing has always been a spine in the economic society. Fishing in the form of farming has been a manufacturing form on the island since 1978 (Åland's Seafood Growers Association, 2017). The landscape of Åland have great fish farming sites and favourable aquaculture conditions. The establishment of the fish farming industry coincided with the radical reduction of wild-caught fish from the Baltic Sea that has occurred the last decades (Wiklund, 2017). The farmed fish is Åland's by far greatest export good, according to Åland's Seafood Growers Association (2017), which writes that the farmers sell fish equivalent to 300 000 human consumption every year[1]. The most commonly farmed fish on Åland is rainbow trout, though whitefish and brown trout are also farmed in smaller amounts (ibid).

Fish farming is a primary production of food, like agriculture and animal husbandry. All food production is associated with emissions of phosphorus and nitrogen. These substances are naturally in the marine environment but have negative effects on the environment if they are high in concentration which cause eutrophication (Åland's Seafood Growers Association, 2017). Since farmed fish is the largest export on Åland, it consequently stands for a significant percentage of the total emissions (37% of the total phosphor and 9% of total nitrogen emissions) (The Government of Åland, 2011). These large shares of emissions from the fish farming industry could be explained by the fact that Åland has a low population and the that it has no major industrial- or agriculture sectors. Seen from a total Baltic Sea perspective however, the extent of the emissions from fish farms are trivial compared to emissions from agriculture, transport and residence (Wiklund, 2017). Nevertheless, because of the high emission rates on Åland, the aquaculture on Åland has been very disputed and faced strict requirements of emission reductions and measures to decrease negative environmental effects that arise because of the production (ibid). The numbers of farms have reduced from 21 in the year of 2000 to only 7 in 2016/2017, as can be seen in Figure 1. The number of employees have reduced under the same years from 90 to 63 (Statistics and Research Åland, 2017a).

[1] Based on average Scandinavian consumption of fish.

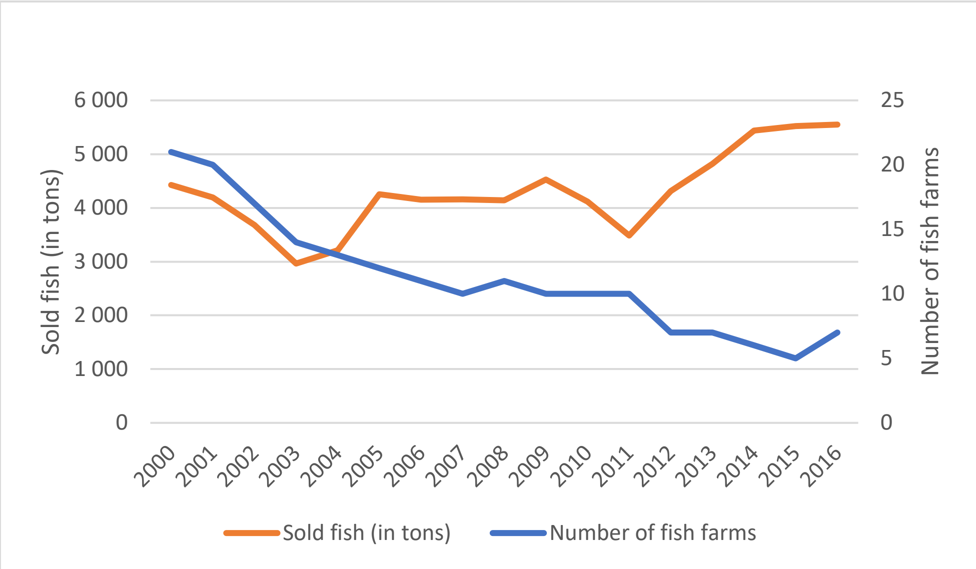


Figure 1. Produced amount of fish and number of fish farmers on Åland. Source: Statistics and Research Åland, 2017a.

2.1.2. Socioeconomic possibilities from fish waste

In the last few years, Åland have established long term goals of improving resource efficiency and become a pioneer in circular economy. This is not least stated with the Government invested "Åland's Sustainable Food Strategy" (Wiklund, 2017). The aim of the strategy is that to through circular economic systems and innovation develop an agriculture that is both climate smart and resource efficient. Several "problems" in the agricultural chain are recognized and debated, among them is the fish farming industry (ibid). It is stated that the industry needs to be seen through other lenses than before to develop a circular business model, were the fish farmers are all closely interlinked to create synergies for each other and for other actors (ibid). A closely related environmental issue recognized in the project is the mineral fertilizer that is commonly used in agriculture on the island today (Wiklund, 2017). It is produced in an energy-intensive and environmentally unfriendly manner.

In line with the strategy of Sustainable Food Strategy is the project "Added value from fish waste" founded by the European Maritime and Fisheries Fund and run by Åland University of Applied Sciences. The project aims at exploring the possibilities and conditions for creating an added socioeconomically value from the residue of fish farming on Åland (Åland University of Applied Sciences, 2017). This thesis is written in collaboration with the project and will contribute by examine the socioeconomic value of making biogas from the fish waste.

There is currently no large-scale biogas production on Åland. Though the potential and possibility of a biogas factory have been topical earlier. Allerborg et al. (2015) performed a study with the purpose of investigating the possibility of an environmentally friendly and cost-effective way of handling biofuel on Åland through digestion. Their investigated biogas production includes all available waste on Åland as substrate. Various alternative technical solutions, with regards to pre-treatment, digestion and reprocessing, are investigated. The study finds that 35 000 tons of substrate of different kinds is available for digestion which can produce 20 GWh (gigawatt-hours) annually. It recommends a digestion process including a pre-treatment with hygienization and an upgrading to vehicle gas. Apart from technical solutions, the analysis comprises an investigation of where a biogas factory is best located, the location is recommended to Svinryggens Deponi in Gottby on Åland.

3. Chapter 3 – Previous Research

There are several evaluations of the socioeconomic impacts of increased biogas in the transport sector. The substrate used to produce biogas varies in the literature. However, despite this, earlier studies overlap regarding the identified socioeconomic impacts of transitions to biofuels.

In a study by Brännlund et al. (2010), the purpose was to estimate the socioeconomic value of making biogas from three identified environmental impacts. The first environmental impact was reduced methane and nitrous oxide (laughing gas) emissions which is identified especially when biogas is produced from manure. The second was reduced particle emissions when biogas replaces gasoline from the transport sector. The third impact was reduced nitrogen leakage when bio fertilizer replaces artificial fertilizer in agriculture. Departing from an example where 20 GWh biogas were produced they found that each impact is very dependent on conditions and assumptions. Reduced methane and nitrous oxide was found to have a large effect on society, between EUR 44 700 and 222 700[2], depending on the price of CO_2 equivalents. The effects from reduced nitrogen ranged between EUR 77 06 – 15 400[3] which were said to be estimated as small in this context. The economic impact of particles was found to be sensitive to assumptions about where the emissions are emitted and what vehicles that use the biogas. A result for the given production of 20 GWh was not presented for particle emission.

Anderson et al. (2016) performed a cost benefit analysis (CBA) of the socioeconomic impacts of a biogas expansion in the Jönköping region. The scenario evaluated in contrast to status quo was a scenario where all waste from food consumption in the region is taken to produce biogas through anaerobic digestion. The three primarily identified societal benefits were analogous with the one found in Brännlund et al. (2010), though this study also included reduced emissions of CO_2 under the assumption that biogas replaces gasoline in the transport sector. The reductions of CO_2 equivalents were monetized based from the Swedish carbon tax. The reduction of emissions from particles was based on estimated willingness to pay to avoid emissions taken by the Swedish Transport Administration. Reduction of leakage of nitrogen was monetized with a willingness to pay method for avoiding eutrophication. Above these effects, they discussed benefits of energy security, employment and possible gains of exports qualitatively. Their result showed that the values were sensitive to assumptions as the net socioeconomic value laid between EUR 4,7 to 21 million per year[4]. These costs excluded benefits from employment which was found to be too uncertain to include.

Another cost benefit analysis has been made by Blidberg et al. (2013) with the objective to examine the socioeconomic viability of an increase in biogas production with reed as substrate in the Kalmar region in Sweden. A sub-objective was to propose areas in the production where there could be improvement potential for increased profitability. The study assumed that the produced biogas would be upgraded and used as vehicle fuel. External effects from reduced CO_2e emissions, reduced nitrogen leakage and reduced particles was included in the analysis. The study concluded that the costs are higher than the benefits of making biogas from reed, since the harvest of reed were expensive alongside the biogas potential from the substrate that showed to be relatively low. However, the authors believe that there is a great potential for technology and efficiency improvement that could lead to profitability in the future.

Tufvesson et al. (2013) have performed a socioeconomic analysis of making biogas from manure. The quantity examined was the whole biogas potential from manure in Sweden which amounts to 3 TWh

[2] SEK 406 000 – 2 023 000 (2010 years prices)
[3] SEK 70 000 – 140 000 (2010 years prices)
[4] SEK 48 to 202 million (2016 years prices)

annually. The purpose was to find net benefits of this production by calculating the environmental impact to produce biogas as well as for the end use of biogas as fossil fuels are replaced as vehicle gas. They found that the net benefits are positive irrespectively of whether the social benefits of exchanging fossil fuels with biofuels are included or not.

Höjgård and Wilhemsson (2012) also performed a socioeconomic analysis of biogas production from manure. The purpose of the study was to use the information available to quantify the net effect on emissions under Swedish conditions and to estimate the societal value of this effect per Nm^3 methane produced. The environmental impacts studied were global warming from GHGs, eutrophication, acidification and adverse effects on human health. This study is a bit different from the earlier mentioned as it does not include societal effects of the end use for biogas, only the production. The result showed that the environmental costs from the production were contradicting, it reduced greenhouse gases and eutrophication which gave positive societal values on the one hand, but it increased local emissions of particular matters and acidifying substances on the other. The net socioeconomic value of the biogas production could not be confirmed to be positive which brought the authors to a discussion of whether the production should be subsidised or taxed. They concluded that it depends on the values chosen for the identified impacts.

Only one study found evaluated the socioeconomic value of producing biogas from an aquatic residue. Stenberg et.al (2013) have performed a study where the production system of cultivation of sea squirts for biogas have were evaluated from an economic and biological perspective. The purpose was to calculate the socioeconomic benefits of cultivating sea squirts for the benefits of purifications of the sea and sell it as biogas. Without consideration for external costs, the results showed negative outcomes on all scenarios (the different scenarios are different quantities of sea squirts, with or without an already existing biogas factory) with the lowest negative balance of EUR 160[5] per kg dry substance of sea squirts. If purification benefits were monetized in accordance with the Swedish Environmental Protection Agency's suggestion of a certification system for nitrogen removal of EUR 5,3[6] per kg of nitrogen, all the scenarios where a biogas factory already exists reach balance. For scenarios including investments in a new factory the break-even production scale of 35 GWh.

The above-mentioned studies have many similarities, especially regarding the identified impacts of biogas production. The ones normally found are reduced greenhouse gases, reduced particle emissions and reduced nutrient leakage to the sea. The studies that has evaluated biogas production from manure also placed large weight on reduced methane emissions. The methods and values they used to monetize each environmental impact differ and the more largely scaled ones (Brännlund et al., 2010, Höjgård and Wilhelmsson, 2012 and Tufvesson et al., 2013) have put larger emphasize on insecurities on the chosen values. However, none of the studies found evaluated the impacts over time or regarded increasing damage cost with increased emission stocks over time. This study will evaluate the socioeconomic value of producing biogas from a waste product over time which can demonstrate the sensitivities of valuation methods and choice of discount rate in a way that has not been done in the literature before.

4. Chapter Four – Conceptual Framework

Fish is an important natural resource on Åland and the fish farming industries are an important source of employment and income. However, the residual from farming could possibly be used in a more

[5] SEK 1516 (2013 years prices)
[6] SEK 50 (2013 years prices)

efficient way then it is currently. The purpose of this analysis is to compare a scenario where fish waste is used for biogas production with status quo by its socioeconomic value. This involves taking the environmental benefits into account that are not priced on regular markets. In economics, a third party detrimental (or beneficial) effect for which no price is extracted is referred to as an externality (Pearce et al., 2006). The concept of externalities has been familiar in economics for a great time but it was Pigou in 1920 who developed the concept of the divergence between private and social cost, the divergence being the value of the externality (ibid). Externalities are a source of market failure, this since the total social price of a product or service is not represented which means that resources in the economy is not optimally allocated. If there is a beneficial externality the market will produce too little of it in relation to the requirements of allocative efficiency, while in the case of a detrimental externality the market will produce more of it than efficiency requires (Perman et al., 2003). The economic literature presents solution to the market failure of externalities in form of for example government intervention such as taxes or subsidies (Pearce et al., 2006). Another solution is to assign suitable property rights where private bargaining between individuals can correct externality problems and lead to efficient allocations (Perman et al., 2003).

In this study, the case of producing biogas and replace fossil fuels, the transition is expected to create a beneficial externality in the economy since pollution is reduced. If no attention is put on this external effect, the value of biogas production will be underestimated and too little biogas will be produced. To account for this market failure, the divergence between private and social cost must be valued and included in the analysis. To evaluate a monetary valuation of an external effect or a public good that is not priced at any existing market involves quantifying two measures; the size of the external effect in physical terms, and the economic value per unit of the physical effect (Pearce et al., 2006).

To examine the socioeconomic value of biogas production, a basis is needed that allows for converting the all values, both external and internal, into the same entity. This lead on to the next section on methodology.

5. Chapter Five – Cost Benefit Analysis

The method applied to answer the research question is a Cost Benefit Analysis (CBA). CBA is an economic method cataloguing impacts as benefits and cost and monetizing them with the purpose of evaluating a net benefit (benefits minus costs) of a studied proposal. The net benefit of the proposal is compared to status quo (business as usual) to give an economically efficient indication for decision-making of public and private sector projects (Boardman et al., 2014).

In a public proposal, the costs and benefits could be seen as positive and negative effects for the society´s utility and the objective is to reach more efficient allocations of society´s resources. The CBA attempts to correct for market failure by including externality effects that are not valued on present markets. The external effects are monetized and treated as arguments in utility functions along with ordinary inputs such as labour, capital and raw materials to get a correct calculation of the projects impacts on society (Perman et al., 2003).

Most projects have costs and benefits that occur over time, first these impacts need to be recognized and then they need to be aggregated and valued in the different years they arise. This is done by discounting future benefits and costs to present benefits and costs (Boardman et al., 2014). Discounting is made for two reasons; one because of the opportunity cost of the resources used in the project, as the resources could be transformed into greater amount of resources in the future. And the other because people are impatient and unsure about the future why they wish to consume now rather than later. Both reasons are consistent with a positive discount rate (ibid).

Once all relevant costs and benefits have been monetized and converted into present value terms they are added to Net Present Value (NPV) (present value of benefits (PVB) minus present value of costs (PVC)) of the project (Hanley and Spash, 1993). The net present value (NPV) is calculated as:

$$NPV = \sum_{t=0}^{n} \frac{(Benefits - Costs)}{(1+r)^t}$$

(Equation 1)

Where (1+r) denotes the discount rate and t is the time (length) that the costs or benefit will last (Boardman et al, 2014). The social rate of time preference, r, can be defined by the Ramsey equation:

$$r = \rho + \eta g$$

(Equation 2)

Where ρ is the pure time preference i.e. a measure of people's impatience, η is the elasticity of the marginal utility of consumption and g is the growth rate. Section 5.3 discusses discount rate further and present the chosen discount rate for this project. The general decision rule of CBA is to perform the proposed policy if the NPV of the project is positive, hence if,

$$NPV > 0$$

(Equation 3)

This is commonly referred to as a net present value test (Perman et al., 2003). If the analysis contains more than one alternative project the alternative with the highest NPV should be selected. If neither of the alternatives have a positive NPV, the status quo is most efficient and should persist (Boardman et al, 2014).

5.1. CBA and theoretical view of welfare

Thus, with the method of CBA, one can determine whether the welfare of society increases or decreases if a project is implemented. Social welfare is then a function of all individuals collected benefit (Boardman et al, 2014). It is quite clear that if the benefit increases for some individuals without reducing it for someone else, implies that welfare in society has increased, it is Pareto efficient (ibid). But such obvious cases are unfortunately rare. Instead, it is usually the case that some people get better if a project is implemented while others get worse. Hence, if the theoretical view of welfare in CBA would be strictly in accordance with Pareto efficiency, hardly any project would pass the NPV test (Perman et al., 2003). Instead, improvements in social welfare is commonly estimated with the "potential Pareto improvement'-test according to which a change is desirable if the gainers could compensate the losers and still be better off than status quo. Observe that actual compensation is not required (ibid). This thesis will regard socioeconomic welfare in accordance with potential Pareto improvement and approve the project if the NPV of the project is above zero.

5.2. CBA and the environment – Non-market valuation

To study environmental economics by using CBA is neither a new nor an uncriticised method. Projects, both private and public, regularly have environmental impacts that does not appear in traditional market valuations (Pearce et al, 2006). CBA can include these externalities by monetizing them alongside ordinary inputs. Economists are challenged with developing techniques on how to best perform non-market valuations so that both environmental improvement and environmental deterioration can be included in CBA. The essential idea of these valuations is to ascertain what the affected individuals collectively would be willing to pay if there were markets for these products or services (Perman et al, 2003). If the project brings an environmental improvement the estimation is based on the affected individuals' willingness to pay (WTP) for the change occurring and if the impact is an environmental deterioration the estimation is based on the individuals' willingness to accept (WTA) for the change occurring (Perman et al., 2003).

The empirical methods to evaluate different non-market priced natural resources or emissions can generally be divided into two approaches; indirect and direct approach. Indirect methods exploit data indirectly by observing people's actual behaviour, why they are also referred to as observed behaviour or revealed preferences (Boardman et al., 2014). Examples of indirect methods used in environmental economics are the travel cost method and the property value method. In the former, it is possible to find out what people actually pay to, for example, get to a recreational area, such as a beach. The individuals' travel expenses can then be the basis for evaluation of water quality (Brännlund et.al, 2003). For the later, different property values can be evaluated when the only difference between the properties are environmental conditions. The difference in value is hence the willingness to pay for the environmental difference (Pearce et al., 2006). Pros for the indirect methods are that they reflect actual behaviour, the cons are mainly that they do not reflect the total value of natural resources, it captures only the use values (Boardman et al., 2014).

Direct methods involve using hypothetical markets trying to uncover people's willingness to pay for a resource. The most used direct method is Contingent Valuation Method (CVM), which is performed by asking the affected population questions about their willingness to pay or to accept for an environmental change (Perman et.al, 2003). It is called contingent because the valuation is contingent on the hypothetical scenario put to respondents. The main problem about the approach is that it uses hypothetical questions which can give rise to bias since the respondents do not answer to their true willingness to pay or accept (ibid).

A commonly criticized component of the CBA is whether these estimations are accurate valuations of the monetary value of the non-market costs and benefits (Pearce et al., 2006). The estimations are particularly questionable for environmental subjects when making judgements of long-term effects because of irreversibilities, risks and uncertainties. According to Perman et al (2003), there are many sceptics that argue that people do not relate to the environment accurately and therefore should not be responsible for valuing it, their valuation is simply wrong, independently of whether the evaluation method is indirect or direct.

Another critique, coming from more ethical grounds, is that it is wrong to believe that environmental consequences only affect humans. These criticizers disapprove that only humans have moral standing (ibid). Though few advocates of CBA argue that individuals' preferences are the only valued judgement relevant and the techniques for non-market evaluation is constantly developing to meet this critique (Pearce et al, 2006).

The techniques that have been used for calculation in this analysis is presented, motivated and debated together with their respective impact in the empiric's section.

5.3. Discounting cost and benefits over time

There is no consensus on an optimal discount rate in CBA, and the choice of rate is of great relevance for the net result. This is particularly true when, as with many projects involving environmental impacts, the time horizon for the NPV test is many years into the future (Perman et.al 2003). For this reason, there is a general emphasise of the importance of performing a sensitivity analysis to test the robustness of the result when differencing the rate of interest (Boardman et al.,2014; Perman et al., 2003; Hanley and Spash, 1993).

The values of ρ, η and g from Equation 1 can be based on results in empirical studies and/or more ethical grounds. The ethical ground refers to future generations and that will value the studied effects, but whose voices are not heard in current day choices (Boardman et al, 2014). The higher the assumed interest rate, the less value does the project value the effects on coming generations. When calculating environmental effects this ethnic ground is stated to be additionally significant (Perman et al, 2003).

There is argumentation for declining discount rate for long projects as the value of an impact increases with time. Groom et al., 2005, for example, argue for such a declining discount rate for to adjust for uncertainty related to environmental long-term problems. However, this study will use a constant discount rate, in line with the recommendation of Boardman et al. (2014) that a declining social discount rate should be used only for projects with impacts beyond 50 years.

In the base case of this study a discount rate of 3,5% is used to discount future effects. This rate is in accordance with ASEKs (Method of analysis and socioeconomic calculation values for the transport sector) (2016) recommendation of discount rate for risk assessment and other environmental effects. In the sensitivity analysis, a lower value of 1% and a higher value of 5% will be used to test the robustness of the project based on assumed discount rate.

When conducting projects where costs and benefits will be generated in the future, the choice of relevant time frame will be of great importance for the final results. The economic time frame chosen should be relevant for the included impacts in terms of context and circumstances (Boardman et al., 2014). The time frame for environmental impacts is greatly discussed among researchers as the societal value of an environmental change is difficult to predict. The total time frame of this study is chosen at 15 years, as the biogas factory is assumed to last in 15 years.

5.4. Data collection and Benefit transfer

It is beyond the extent of this study to carry out own evaluation studies. Instead the method of benefit transfer has been used for the evaluation studies, i.e. the quantification and monetization of effects will use values derived from earlier valuation studies. The rationale for using this method is that there are sufficient similarities between the earlier valuation and the valuation made in this study (Boardman et al., 2014). To find relevant values for this project, a literature review has been made and will be presented for each valuation. Statistics about the amount of fish waste and the price of different allocations used in the reference scenario is received from the project "Added value from fish waste".

6. Chapter Six – Studied Scenarios

The following chapter initially motivates the choice of a biogas scenario. Section 5.1 describes the Reference Scenario, i.e. today's management of fish waste and today's use of fossil vehicle fuels on Åland. This is the reference scenario against which socioeconomic impacts of producing biogas from fish waste is compared. The comparison starts after the slaughter process at the fish farms. In section 5.2, the analysed biogas scenario is presented with important premises and assumptions.

6.1. Status Quo – Reference Scenario

An important factor for whether biogas production from waste is profitable is the price of the substrate. Some substrates types bring a reception fee for substrate owners. These are usually those waste products that the owner must pay to get rid of elsewhere, that is, if the opportunity cost is greater than zero (Vestman et al, 2014)

The six fish farms on Åland together produce 2 540 ton of fish waste per year from their production (Åland University of Applied Sciences, 2017). A fraction of this waste (70 ton) is used today to produce biodiesel in a small-scale production at the farm (ibid). The cost of this production is not accessible and difficult to generalize in such a small scale why they have been removed from the analysis. Left to analyse is 2480 ton of fish waste per year.

Most of this waste (about 83%, see Table 1) is sold as food to mink farms in mainland Finland (Åland University of Applied Sciences, 2017). This waste is ensiled with formic acid to preserve nutrient content in the fish, after that the waste packaged, sold and sent to mink farms in Ostrobothnia in Finland. The price that the farms receive for the ensiled waste is EUR 200 per ton. Some waste (280 ton or 11%) is sold as fish feed today to a factory in mainland Finland. The estimated price that the farmer receives from this is the same (EUR 200 per ton) as selling it to the mink farms (ibid). Irrespectively of whether it is sold as fish feed or to the mink farms, the buyers pay the cost of transportation. Another part of the waste is composted (140 ton or 5,6%) (ibid). This brings a benefit as it can be used as manure. The value of composting is set with at EUR 10 as the price of fertilizer on Åland is expected to be sold at that price (Wiklund, 2016).

The costs of the treatments of the three different allocations is so low that it will be considered a zero cost in the analysis. Though the benefits will be considered opportunity costs when evaluating the Net Present Value of the alternative biogas scenario.

Table 1. Total revenue from reference scenario

Use	Amount per year	At price:	Total revenue EUR per year
Feed production	2050 ton	EUR 200/ton	410 000
Sell as fish feed	280 ton	EUR 200/ton	56 000
Compost	140 ton	EUR 10/ton	1 400
Total	2 480 ton		467 400

Source: Åland University of Applied Sciences (2017)

6.2. Biogas Scenario

6.2.1. Production of biogas from fish waste

There is no biogas plant on Åland today why the scenario includes building a new biogas plant on the island where all the fish waste from the seven studied farms are transported. The biogas is produced at the plant through anaerobic digestion. Anaerobic digestion is a collection of processes by which micro-organisms breakdown biodegradable materials in the absence of air (Yuvaraj et al., 2016). The scenario assumes that the process occurs under mesophilic conditions (at a process temperature around 37 ° C) which is the most common solution in Sweden (Swedish Energy Agency, 2012). Since fish waste is an animal by-product, it must be hygienized before digestion according to EU-law (Stenberg et al., 2013).

After the digestion process, a rest product remains consisting of hard biodegradable organic matter, inert inorganic material, nutrients, microbes and water. Due to the high nutritional content, this rest product is used as a fertilizer on farmland and is then called bio fertilizer (Brännlund et al., 2010). In

99% of the larger biogas factories in Sweden this rest product is used as fertilizer. In this study, the produced bio fertilizer is to be sold and used by famers on Åland and replace mineral fertilizer.

6.2.2. End use of biogas

Vehicle gas is the scope for biogas in the Nordic countries that produce the greatest revenue due to higher willingness to pay of the end customer (Brännlund et al., 2010). Additionally, vehicle gas is the fuel on the market with the lowest carbon content and is therefore one of the fuels that has the absolute least climate and environmental impact. And in addition to that, biogas-powered vehicles provide reduced emissions of particles. This study will assume that the biogas is used in passenger cars replacing gasoline. The upgrading is assumed to be done using a water washing technique which is used in most Swedish biogas factories (Edström et al, 2008).

According to (Swedish Energy Agency, 2016a) the price of biogas must be about 20% lower than the price of fossil fuels for demand to exist on the market. Later, as can be seen in section 6.3.1, the price expected at the pump is in accordance with ratio. However, based from an economic theory, the demand assumption is not uncomplicated for two reasons both related to demand theory.

The first one concerns whether it is reasonable to assume full demand for the produced biogas on Åland. It is an essential assumption to argue for since socioeconomic profitability is not thinkable if the produced biogas is neither sold nor used. Sweden and Finland have the largest share of renewable resources in their transport sectors in all of Europe (Swedish Transport Administration, 2017). In Sweden, 63% of the produced biogas goes to the transport sector (Swedish Energy Agency, 2013) and the demand for the vehicle gas is larger than the production in Sweden according to several sources (Vestman et al, 2014; Swedish Environmental Protection Agency, 2012). In this context, the assumption of full demand for the produced biogas is expected to hold.

The second reason for doubt it that this demand for biogas might not substitute the demand for fossil fuels. In theory, it is possible that this demand is added on top of the fossil fuel demand causing a kind of “rebound effect”. Rebound effects are generally discussed in environmental economics as an explanation for empirical findings where the reduction in total energy use have not corresponded with the expected as energy efficiency increases. “Buy a more fuel-efficient car, drive more”, is according to Gillingham et al. (2014) the most well-known intuition for the rebound effect, and the quote is very close to the subject of this thesis. A rebound effect in this case would be that the reduction in emissions is lower than expected because the increased supply of biogas at a low price increased total demand. A worst-case scenario would be a “backfire” if the project led to an overall increase in emissions.

However, the probability of a rebound effect is considered low in this case. When the Swedish Environmental Research Institute (IVL) (Rydberg et al., 2010) investigated the market conditions for biogas they performed a survey analysis which concluded that the increase in demand for biogas would be at the expense of fossil cars, hence no rebound effect. The assumption is neither rare in the literature, both Tufvesson et al. (2013), Stenberg et al. (2013) and Blidberg et al. (2013) make the same assumption. With support from this, this study will do the same.

In conclusion, in the studied biogas scenario, the gas is upgraded to vehicle gas after the digestion process. Thereafter it is distributed to a gas station that is assumed to be in connection to the biogas factory. The gas is bought and used by the habitants on Åland. Left from the production of biogas is a rest product in form of bio fertilizer that is bought and used by farmers on Åland.

6.2.3. Biogas potential

Different substrates have different biogas yield per kg substrate. To analyse the socioeconomic value of biogas production from fish waste it is essential to know how much biogas that can be produced from the given amount of fish waste. According to the digestion tests presented in (Carlsson and Uldal, 2009) one ton of fish waste results in 537 Nm^3 biogas (1 Nm^3 equals approximately 0.8 kg). In a more recent digestion test by (Ekengren et al, 2015) the test show a bit lower biogas yield from one ton of fish waste, namely 434 Nm^3 biogas. The digestion test by Ekengren et al. (2015) is performed in the same way as the production process is assumed to in the biogas scenario presented in section 5.2. This study will use the numbers of the newer test, since the test is closer to date and to avoid overestimating the biogas yield. 1 Nm^3 of biogas is equivalent to 9,67 kWh and one kWh is equal to 3,6 mega joule (MJ) (Biogasportalen, 2015). In accordance with these numbers the table below shows the quantity of biogas that can be produced from the fish waste within the project.

Table 2. Quantifying biogas yield

Fish waste	Nm^3 per ton	Total Nm^3	Total kg	Total kWh	Total GWh	Total MJ
2480 ton	434	1 076 320	861 056	10 408 014	10,40	37 468 852

6.2.4. Identified impacts

The effects on society's utility from using fish waste as biogas is identified in the figure below that portraits the value chain of the biogas scenario. All costs are marked with a minus sign and all benefits are marked with a plus. All impacts are quantified and monetized in detail in chapter six.

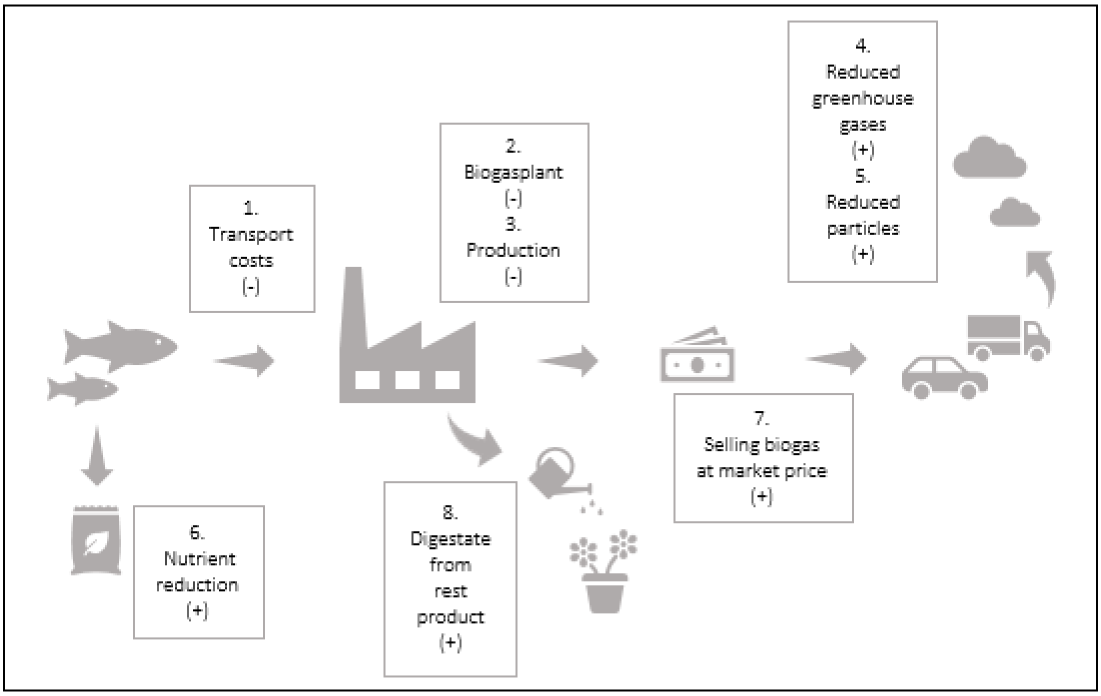


Figure 2 – Identified Impacts of biogas scenario

After the slaughter process of the fish at the fish farms, the waste in the biogas scenario must be transported to the site of the assumed biogas factory, which entails a transportation cost *(1)*. At the factory, the production of biogas through anaerobic digestion is associated with a unit cost for every ton produced. This cost must include investment costs for a biogas factory constructed on Åland *(2and3)*. When the vehicle gas is produced, it is sold at a market price which brings a revenue *(7)*. Switching from fossil vehicle fuel to biogas brings environmental benefits to society in form of reduced greenhouse gases and reduced particles *(4+5)*. Lastly, the production of biogas leave a waste product that can be used as a digestate (biofertilizer) to fertilize farmland, this brings a benefit when the digestate is sold *(8)*. The reduction of artificial fertilizer reduces nitrogen leakage which is served as an environmental benefit (*9)*. Additionally, positive employment factors are identified as an effect of the project.

7. Chapter Seven – Socioeconomic analysis

As the impacts of the biogas scenario have been identified this chapter continues with the socioeconomic analysis. The quantification, meaning the scenarios physical impact in appropriate unit is expressed and motivated. Thereafter, the value of the impact is debated and selected monetary values are presented. Initially the costs associated with the biogas scenario will be presented. Subsequently, the environmental benefits will be considered divided into three categories. Thereafter, the marketable revenues from the biogas scenario will be considered.

The socioeconomic analysis will be based on the assumed production potential presented in the earlier section. All costs are in EURO (EUR) and are expressed in base year 2016 to correct for shorter fluctuations during 2017. When prices are converted, the yearly average exchange rate of 2016 have been used presented by the Swedish Central Bank[7]. One base case evaluation will later be presented in the result section. In the following sensitivity analysis, several adjustments of assumptions and used values are presented as alternative perceptions of the result. The sensitivity analysis also includes a comparison between the best and the worst scenarios which is a common practice among CBA studies (Pearce et al., 2006). It presents the highest and the lowest values for every impact that's been found in the literature to illustrate the most extreme values. The term can be confusing, but the best case is the best possible case for the studied NPV.

7.1. Costs

7.1.1. Transportation costs

Within the biogas scenario all fish farms transport their residue to a shared biogas plant. The transportation there is associated with a cost that must be considered. In the study by Allerborg et al. (2015) they evaluate the best location for a biogas plant on Åland and reach the conclusion that the landfill Svinryggens Deponi in Gottby would be the most optimal. The location can be found in Figure 3 below.

If the location of the factory had been chosen only by economic reasons concerning transport from the studied fish farms the location might have been different. Though the study comprises several other factors such as the distance to Mariehamn where the demand for a biogas fuel station is assumed to be greatest, the transportation distances to collect waste and provide bio fertilizer, any excavation work to fit with the facility and how densely populated it is around the area to minimize the odour nuisance of neighbours.

[7] Where, 1 EUR = 9,4704 SEK and 1 SEK = 0,10559 EUR

The study by Allerborg et al. (2015) include multiple substrates why they evaluate the distance from many more industries, however, the location in Gottby seems best even for a plant collecting substrate from only fish farms. The costs for transportation will therefore be evaluated by the distance from the fish farms to Svinryggens Deponi.

Holgersson (2011) measure transportation costs per MWh (megawatt-hours) and emphasize an important remark on the unit of which the study presents its costs. Different substrates have different energy content why it is not generalizable to apply this cost for every substrate. One example is that you can transport animal residues waste more than 200 km at about the same cost as transporting manure 20 km if you measure it by km per MWh. If examining studies with different substrate is it therefor crucial to use another unit, like cost per ton transported ton.

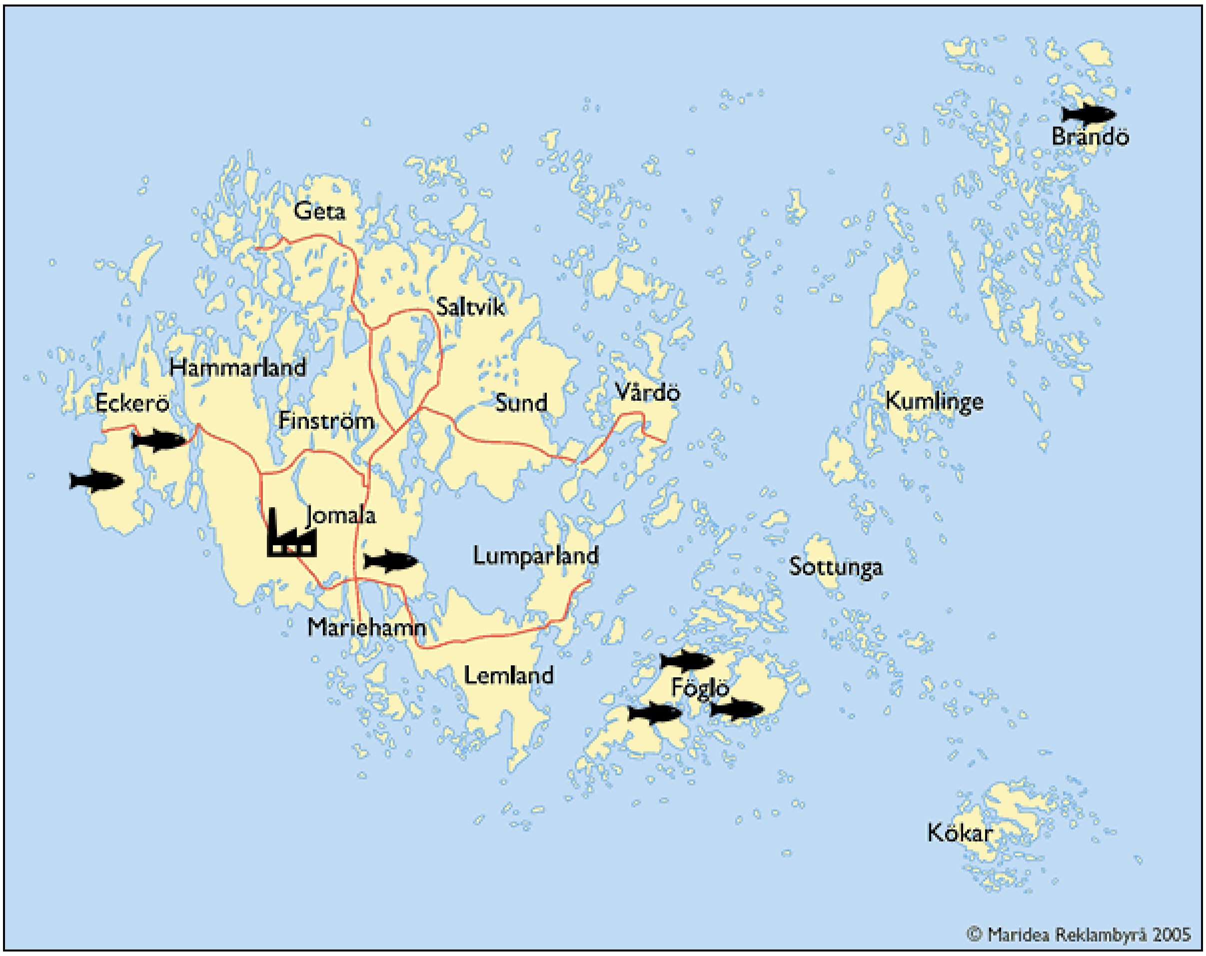


Figure 3. Map of Åland with the fish farms and the assigned biogas plant. Every fish on the map represents a farm. The industry picture is the location of Svinryggens deponi. Source: Own graphics with data from Allerborg et al. (2015).

For this study, the unit of "kilometre transported ton" have been calculated as tons of fish waste per year times the distance to biogas plant for every fish farm. This since the farms produce different amount of fish waste and are differently located from the intended location of the biogas plant. An alternative method would be to measure the average distance that the waste need to be transported from the farms to the plant it would not be as precise. The total kilometre transported ton is 114 820 but since the transport mean needs to go two way the number is doubled. See Table 2 below for calculation.

Table 2. Quantification of kilometre transported ton

Fishfarm	Fishwaste per year in tons	Distance to biogas plant in km	Kilometre transported ton
1	280	117 * 2	65 520
2	120	53 * 2	12 720
3	480	44 * 2	42 240
4	1000	44 * 2	88 000
5	140	37 * 2	10 360
6	450	12 * 2	10 800
Total	2480		229 640

Monetization of transportation

The cost of transporting the substrate depends on several things, for example how large every load is and the transportation method. Concerning transportation mean, it is considered least costly to transport waste in trucks and most expensive to transport it in pipelines (Holgersson, 2011). It is also costlier to transport substrate in liquid form since it requires a tank.

Stenberg et al. (2013) calculate a cost of transport of sea squirts both my trucks and cargo ships, they include both loading and unloading at the biogas factory and even washing the vehicle and measure it in cost per transported ton. The cost they use is EUR 0,14 per ton transported km.

Another study by Johansson and Nilsson (2007) evaluate transportation costs for the substrate manure but emphasise that their result could be generalised to other substrates as well. They evaluate transport costs at different sizes of loads where smaller loads are more expensive per ton transported kilometre. If the load capacity is set to 40 tons the cost is EUR 0,07 per ton transported km but if the load is only 14 tons the costs are set at EUR 0,16. This cost is evaluated for waste of solid form.

Selected Values

This study will use transport costs from Johansson and Nilsson (2007). The higher cost of EUR 0,16 will be used since it is reasonable to believe that many of the loads from the farms will be rather small. Also, it can correct for the fact that some of the waste needs to be transported by boat. The total cost of transport is calculated at EUR 36 742 per year.

7.1.2. Production costs

The most essential cost for the biogas scenario is the cost of producing biogas. This cost is predominantly measured in price per kWh in the literature. Hence, to monetize the costs of producing biogas from fish waste in this study, a function where costs depend on the produced unit (kWh) is desirable. Finding generalizable key costs for producing biogas is difficult since almost no system is the same. The type of substrate affects the costs as some need pre-treatment in form of hygienization (sterilisation). Substrates containing animal residues (such as fish waste) must be sterilised by to avoid contamination, this is legal in the EU (Stenberg et al., 2013).

Another important cost factor is the scale as the production suffers great scale benefits (Carlsson and Uldal, 2009). The scale of biogas production is relatively low in this study compared to the ones normally calculated for in studies of biogas production. However, there is great potential that other substrates could be digested together with the fish waste in the future which could potentially lower the costs ahead. Assumingly the total biogas produced in Allerborg et al. (2015) (20 GWh) could be added to the production. The base case in the result section will assume that only fish waste will be digested, though in the sensitivity analysis results are presented in the possible scenario of increased production scale where total production amounts to 30 GWh.

Monetization of production costs

To find reasonable costs for biogas production a literature review has been made. The reviewed literature studies different substrates for production but they all use the same digestion process. The result of the review is found in Table 3 below.

For clarity, the costs of the production are presented in sections in line with the production stages. The first cost component is raw gas production. Raw gas production is often calculated as a sum of investment costs and operating costs, though the approach in this study is to separate them in the extent it is possible. Hygienization is also a part of raw gas production why costs of this is included in the first component.

The second component is upgrading and compression which is made for the purpose of vehicle gas. The biogas must be upgraded which means that carbon dioxide and various impurities are removed. After the upgrade, the biogas has a methane content of 95-99%. (Edström et al, 2008). Water washing is the most common technique for upgrading biogas, it is used in 70% of the Swedish biogas factories and is said to have advantages in smaller scaled productions why this study will focus on costs for this technique (ibid). The last component is finally distribution of final product and costs for managing a fuel station. The gas station on Åland is assumed to be in connection to the biogas factory, though still costs must be included for distribution of gas to the station and for operating the station.

Table 3. Biogas production cost

Study	Investment EUR/kWh	Operating EUR/kWh	Raw gas EUR/kWh	Upgrading and compression EUR/kWh	Distribution and gas station EUR/kWh	Total cost EUR/kwh	Comments
Edström et al. (2008)	Not specified[a]	Not specified [a]	0,05	Not included[b]	Not included[b]	0,05	Substrate: manure
Lantz and Björnsson (2010)	0,02						Substrate: manure
Stenberg et al. (2013) 10 Gwh production annually	0,04 (Total investment is EUR 0,47 per kWh[c])	0,03	0,07	Not specified[a]	Not included[b]	0,07	Substrate: sea squirts Calculating a biogas factory of 10 GWh per year.
Stenberg et al. (2013) 30 Gwh production annually	0,03 (Total investment is EUR 0,29 per kWh[d])	0,02	0,05	Not specified[a]	Not included[b]	0,05	Substrate: sea squirts Calculating a biogas factory of 30 GWh per year.
Vestman et al. (2014) (Average)	Not specified [a]	Not specified [a]	0,09	0,03	0,024	0,144	Substate: household waste. Studies nine Swedish biogas factories.
Vestman et al. (2014) (Median)	Not specified [a]	Not specified [a]	0,06	0,028	0,019	0,107	Substate: household waste. Studies nine Swedish biogas

							factories.
Blidberg et al. (2014)	Not specified [a]	Not specified [a]	0,032	0,016	Not included[b]	0,048	Substrate: reed. Does not include hygienization.
Swedish Energy Agency (2010)	Not specified [a]	Not specified [a]	0,05	0,014	0,03	0,094	Substrate: Household waste, energy crops, manure, animal waste.

[a] The cost is included in the calculation, but not specified separately
[b] The production does not include this component nor its cost
[c] Total investment cost needed for biogas factory is EUR 0,47 times annual kWh
[d] Total investment cost needed for biogas factory is EUR 0,29 times annual kWh

The study by Vestman et al. studies nine different biogas factories in Sweden and present both a median cost and an average cost of these factories. The investment costs are included in the raw gas production in the study. The study from Swedish Energy Agency also present results as a range, the numbers presented in the table are the average costs presented. The study by Stenberg et al. present costs for two different production scales, one lower at 10 GWh produced biogas annually and one higher at 30 GWh annually.

Selected Values

Clearly the cost of producing biogas is greatly dependent on the conditions for production. The values selected for this study are taken from Stenberg et al. (2013), which is motivated by the fact that the study evaluates digestion from an aquatic substrate and because it calculates costs from a biogas factory of a corresponding production scale (10 GWh per year). The needed investment is reported to be EUR 0,47 per annual kWh resulting in a total investment cost of EUR 4,89 Million for this project (the price per kWh is multiplied with total kWh per year to get the total investment cost).

The cost of capital from the investment is computed by the basic principle that the investment cost is to be equally distributed for every year (ASEK, 2016). It is annuity based on an amortization period of 15 years and an interest rate of 3,5%. The cost of capital is therefore EUR 548 767 per year. Per kWh that is EUR 0,053 annually. The operating cost is 0,03 per kWh. Total cost of raw gas and upgrading is therefore calculated at EUR 0,083 per kWh.

Costs for distribution/operation of the gas station is not included in Stenberg et al. (2013), why this post is instead taken from the average cost presented by Vestman et al. (2014) at EUR 0,024 per kWh. Total cost of production is hence set to 0,11 per kWh, giving an annual cost of EUR 1 144 882.

Since the range of production cost is so great among the literature, the sensitivity analysis will include a graph analysing cost from the lowest to the highest reported cost to bring a discussion of break-even price. The sensitivity analysis will also include a discussion of the production cost if the production scale could be increased to 30 GWh per year. For the best and worst case analysis, the values chosen are EUR 0,07 (lowest possible cost from Stenberg et al. 2013 (EUR 0,05 per kWh) and distribution cost from Vestman et al., 2014 (EUR 0,019 per kWh)) and EUR 0,144 (average cost from Vestman et. al 2014).

7.2. Environmental Benefits

7.2.1. Reduced environmental impacts of greenhouse gases

Biogas production from a rest product (such as fish waste) is assumed to affect the net amount of emissions of greenhouse gases (GHGs) in contrast to the status quo scenario (Brännlund et al. 2003). The first being that biogas can replace gasoline and diesel in the transport sector and thus reduce the emissions of GHGs from burning fossil fuels. Vehicle gas is the fuel on the market with the lowest carbon content and is therefore one of the fuels that has the absolute least climate and environmental impact (Börjesson et al., 2010).

This environmental impact is associated with a value for society that is not priced on the market. The following section will give an insight to the extensive discussion on how greenhouse gases can be valued. Yet, the first part is to quantify the reduction in emissions from the studied scenario.

Quantification of greenhouse gases

This section will quantify the reduced amounts of CO_2 equivalents (CO_2e) from the biogas scenario in contrast to the Reference scenario. CO_2e is a measure of greenhouse gas emissions that consider that different gases have different capabilities to contribute to greenhouse effect and global warming. Expressing the emissions of a certain greenhouse gas in CO_2e indicate how much carbon dioxide would have to be released to give the same effect on the climate (Pearce et al., 2006).

To evaluate the quantity of reduced emissions from replacing gasoline in the biogas scenario the emissions per unit of biogas and fossil fuel needs to be compared. For this emission values from different vehicle fuels reported by the Swedish Energy Agency will be used. The unit the fuels are compared by is CO_2 equivalents (CO_2e) per megajoule (MJ). The measures include both emissions both from using the fuel in vehicles as well as emissions generated from the whole production life cycle, such as emissions from cultivation, processing and transportation and distribution of raw materials and finished fuel (STEMFS, 2011). Hence, this includes environmental costs of emissions from the above sections 6.1.1 *Transport Costs* and 6.1.2 *Production Costs*.

This way of comparing the scenarios full production chain is important since the emissions of fossil fuels are majorly emitted when they are burned in the car but the emissions of biogas are emitted during the production. If the fuels were to be compared solemnly from emissions of cars, the reduction of emissions when exchanging fossil fuels with biogas would be overestimated. (STEMFS, 2011; Swedish Energy Agency, 2015).

The emission for fossil vehicle fuel is set to be 83,8 g CO_2e per MJ which will be used as reference value. The emission value for biogas is 23,6 g CO_2e per MJ which is 71,8% lower than the fossil fuel value. The assumption that every unit of biogas produced will replace fossil fuels is equivalent to a reduction of 60,2 g CO_2e per MJ, see also Table 3 below.

Table 3. Quantification of reduced CO_2e emissions

Reduced CO_2e per MJ Biogas	MJ of biogas produced	Reduced kg of CO_2e
60,2	37 468 852	2 310 196 kg

Monetization of greenhouse emissions

To estimate the benefit of a given reduction in emissions of greenhouse gases there are several approaches and the range of price per unit of emission between the approaches are significant. This section will firstly present an overview of some of these methods and secondly present and motivate the values used for this study.

There are great uncertainties concerning the impact of climate change and greenhouse gases (GHG), especially in the long run. Partly because greenhouse gas emissions are global in nature and partly because the consequences often are not felt until much later (Swedish Transport Administration, 2016). Apart from global effects impacts are also felt locally and regionally and often divided and measured from these different areas (ibid).

The most straightforward method to monetize CO_2 would be to value the marginal damage of a unit of emission including non-market impacts on the environment and human health. This approach is commonly denoted as the Social Cost of Carbon (SCC) (Pearce et el, 2006; Stern, 2007). Even though it is confirmed by many that the SCC is the fundamentally correct valuation principle from a strictly welfare perspective, it is also realised that it is very difficult, or even impossible, to calculate because of uncertainties for future effects (Swedish Transport Administration, 2016; Pearce et al, 2006; Stern, 2007).

A large number of studies have been made to value the SCC. The Stern report (2006) is probably the most known. The report became widely discussed as it included significantly more of the long-term effects that are difficult to quantify. As of costs the report found one lower value corresponding to EUR 0,07 and a higher one, included added climatic effects, at EUR 0,25 (Stern, 2006; Stern et al., 2007).

Tol (2008) presents an overview of 47 studied with 211 different estimates of the SCC. The study found an average value of EUR 0,02. The study also finds that the price reported by Stern is an outlier with a price far above most others. The average value from Tol is used both in Brännlund et al. (2010) and Tufvesson et al. (2013) as their worst-case scenario of SCC in their socioeconomic studies.

An alternative valuation method is to use the Shadow Price of Carbon (SPC). One possible way to operationalize the SPC method is to calculate the "price" on greenhouse gases, based on politically set reduction targets, that will achieve the set goal (Brännlund et al., 2010). This is the method used by the ASEK-report (Swedish Transport Administration, 2016) as they use the Swedish Carbon tax at EUR 0,12 to evaluate the marginal cost of greenhouse emissions. For sensitivity analysis, they recommend the higher value of EUR 0,35. This method has the benefit that it can be perceived as an expression of a political objective concerning reduction of CO_2 which is already set in national currency (ibid). A problem with this valuation is that different sectors have different goals and different costs of reducing emissions to achieve that goal. When using the same price over all sectors this can result in the socioeconomic problem that the reductions are distributed in a non-cost-effective way (Tufvesson et al., 2013).

Another related socioeconomic problem with the ASEK- values is presented in (Brännlund et al., 2010) that states that if the SPC method practiced in Sweden use a higher value than the actual cost of SCC this would result in more reductions than economically optimal.

Another problem arising from this valuation is that greenhouse emissions are global in its effect. That means that a marginal increase in emissions in Sweden must give rise to a cost equal to EUR 0,15 globally for the cost to be accurate (Brännlund et al., 2010). This problem could be portrayed by the cost decided by Extern-E, a European project meant to find the shadow price of achieving the reduction goal of the Kyoto-protocol, which was found between EUR 0,005 – EUR 0,02 (Tufvesson et al., 2013). Hence, the marginal cost of emissions chosen with the SPC-method are dependent on the level of ambition of the emission goal.

Yet another method for valuing the shadow price for CO_2 emissions is to use the price of emission allowances. However, this requires a well-functioning market, which has been difficult to achieve (Swedish Transport Administration, 2016). The amount of emissions allowances is politically chosen and has been set quite high until 2020 which has forced the price down, in 2016 the price was EUR 0,01 (ibid). According to the Swedish Transport Administration, 2016 the amount is set too high and the price too low to be in line with what is expected to be a sustainable level of carbon dioxide emissions to keep the 2-degree goal. The study by Höjgård and Wilhelmsson (2012) combine these two mentioned ways of valuing SPC by calculation with a lower value from the emission allowances at EUR 0,007 (the price of an allowance in 2012) and a higher value of EUR 0,13 based on the Swedish carbon tax at that time.

When calculating the price of carbon, it is vital to discuss a non-constant, convex, marginal damage cost. Increased marginal cost are discussed as a mean of climate sensitivity and uncertainty of future effects. As the marginal costs of carbon is expected to increase with increased stock of carbon in the atmosphere it common to use social cost estimates that increase through time (Clarkson and Deyes, 2002; Maibach et al., 2008). However, since the damage cost of carbon is difficult to estimate in the first place, it is even more difficult to estimate as an increasing function based on future environmental effects. It does not therefore exist a general method for incorporating the rising social cost (Clarkson and Deyes).

Implementing convex cost functions is significant from both the SCC and the SPC approach. For the SCC, the reference scenario is central. A reference scenario involving relatively large future emissions implies an increased value of marginal SCC as the damage cost of emitting one more unit increases with large emissions (Brännlund et al., 2010). In Clarkson and Deyes (2002) for example, this is made by increasing the price per ton of carbon by EUR 1,74 annually (GBP 1 in 2002 prices).

For the SPC method, increased costs of carbon are associated with the increased cost of achieving a set emission goal. In the ASEK-report, increased marginal cost is adjusted for by the recommendation of increasing the price of carbon by 1,5% annually as the political valuation increases (Swedish Transport Administration, 2016).

Another way to increase the cost of carbon over time is to use a declining discount rate. With a declining interest rate the costs of carbon will increase in the future which portrays the increased damage cost with increased stocks (Clarkson and Deyes, 2002). This is advocated by many studies (see for example Pearce et al, 2006 and Groom et al, 2005). The socioeconomic effect from this is tested in Guo et al., (2006) who found that the cost of carbon will increase with falling discount rate by as little as 10% or by as much as a factor of 40, depending on the scenario selected.

As the discussion so far in particular concerns the cost of carbon it is appropriate to mention other emissions that are included in the CO_2 equivalents calculated in the quantification section. One of them is methane gas which has higher environmental damage per kg. The damage from methane is 25 times as much that of CO_2 which means that 1 kg of methane is equivalent to 25 CO_2e (Swedish Transport Administration, 2016). However, it is expected to have a shorter lifetime in the atmosphere compared to CO_2 why its emissions do not damage the air as long (Clarkson and Deyes, 2002). Calculating these emissions with the same price and discounting them equally does not give a precise estimate. When evaluating the benefit of exchanging fossil fuels with biogas, a quantification where emission types are evaluated separately and where different impacts over time have been considered is more precise. However, in earlier literature it is not uncommon to translate all emissions into CO_2e, see for example Brännlund et al (2010) and Tufvesson et al (2013).

Selected Values

Because of the uncertainties for estimating the SCC of carbon this study will use the shadow price recommended price by the ASEK-report. The price estimated is EUR 0,12 per kg of CO_2e emissions (see Table 4 below for total benefits from reduced greenhouse emissions).

Since the values presented in the literature is so diverse, the sensitivity analysis will test the result with an interval of values between the average price found in Tol (2008) at EUR 0,02 to the higher price of EUR 0,35 recommended for sensitivity analysis in the ASEK-report (Swedish Transport Administration, 2016). The sensitivity analysis will also include an estimation of carbon cost effects with increased damage cost by using the method from ASEK where the price is increased by 1,5% annually. The emissions are presented as CO_2e which means that the result will not include an adjustment to different emissions lifetimes, précising the long-term costs of divided emissions is instead recommended for future studies.

Table 4. Total benefit from reduced greenhouse emissions

Reduction of kg CO_2e	Price per kg CO_2e in EUR	Total benefit in EUR
2 310 196	0,12	277 223

7.2.2. Reduced environmental impacts of particles

Emissions of particles are increasingly noted as a serious environmental problem, not least because of severe effects on airways and lungs. The particles referred to are exhaust gas particles, $PM_{2,5}$[8]. They are small particles that give negative health effects when inhaled. The particle emissions from biogas is lower than fossil fuels in vehicles, with the assumption that the produced biogas in the biogas scenario replaces fossil fuels this would lead to a social benefit of reduced number of particles (Brännlund et al, 2010). Particle emissions, unlike global greenhouse gas emissions, are more local in its effect why assumption of where the emission take place is essential (ibid). The impact of a given emission will differ depending on the concentration of the pollutant in the air and the number of persons affected. This implies that emissions in an area where the concentration is high will result in higher costs for society as well as emissions at a location with a high population concentration (Höjgård and Wilhelmsson, 2012). The estimations of the socioeconomic value of reduced emissions from the biogas scenario will therefore include aspects of population affected as well as a discussion of increased marginal costs from increased stocks.

To illustrate the path between particle emissions and the monetary valuation of these, the Impact chain approach (Impact Pathway Approach (IPA), is commonly used (Brännlund et al., 2010; Maibach et al., 2008). This approach is based on deriving the impact of the emissions and then giving these effects a monetary value. It is regularly used when evaluating externalities from noise and air pollution (Forslund et al., 2007). An alternative approach would be to go from nation-aggregated emissions of particles and scale them down to local levels and calculate effects. However, according to Forslund et al. (2007) this could underestimate the impacts on pollution on local level why the IPA approach will be used in this study.

The calculation in the Pathway Approach is done in two steps, the first step is to quantify the airborne pollutants. This requires emission factors that measure how many kg of particles a source of emissions causes (Brännlund et al., 2010; Forslund et al., 2007). This is done in the coming section of *Quantification of particle emission*. The second step is to estimate the spread of the emission and

[8] $PM_{2,5}$ are all particles with a diameter less than 2.5 micrometres. It can range from soot to road dust - it is defined by the particle size.

finally multiply the quantity of particles with the cost of its pollution. The valuation of emission costs is commonly done by calculating the cost of the pollutant in form of health impacts and willingness to pay to avoid the emissions (Forslund et al., 2007). A more in depth discussion of this is found in the section *Monetization of particle emission.*

Quantification of particle emission

The biogas scenario causes particle emissions through the production process, however these are compensated with even greater reductions of particles when biogas replaces fossil fuels in vehicles. To evaluate the net reduction in particles from the biogas scenario the particle emissions from biogas and fossil fuels must be compared.

In Brännlund et al. (2010) particle emissions from both biogas and fossil fuels are reported and the numbers will be used in this analysis. The emissions include the whole production cycle from transport of the substrate to all stages of the production (hence, include environmental costs from both 6.1.1 Transport costs and 6.1.2 Costs of production). The emissions are 4,8 mg for fossil fuel and 3,2 mg for biogas per MJ resulting in a reduced number of 1,6 mg per MJ of biogas. An important notice is that for the reference case (fossil fuel), the main emissions occur when using the fuel while the emissions of particles for biogas are more evenly distributed throughout the production life cycle (Brännlund et al., 2010). As with the calculations of greenhouse gases it is therefore important to compare the fuels emissions from the whole life cycle. Table 5 below shows net reductions of particle emissions from the project.

Table 5. Quantification of particles reduced

MJ of produced biogas	Reduced mg particles per MJ	Total mg particles reduced	Total kg particles reduced
37 468 852	1,6	59 950 163	59,95

Monetization of particle emissions

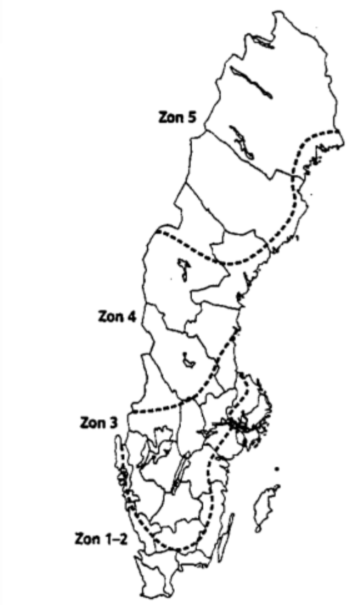
Several different calculations are used in the literature to evaluate the cost of particle emissions. The ASEK 6.0 report by the Swedish Transport Administration (2016) evaluate health effects resulting from particle emissions by evaluating shortened lifetime (mortality) and increased morbidity (Swedish Transport Administration, 2016). The method used to evaluate mortality is to convert the value of a statistical life (VSL) to the value of a life year lost, VOLY. ASEK applies the VSL value used for traffic accidents, which amounts to EUR 2,2 million. The calculation of the air pollution's effects is in price/ kg emission and uses a two-step method. Step one is to calculate the “exposure units” per kg emissions in the area studied. This is made with the equation:

$$Exposure: 0{,}029 * Fv * B * 0{,}5$$

(Equation 4.)

F_v = Ventilation factor for the area
B = Area population

The ventilation factor is the exposure per person and kilo emissions in the studied area. In Figure 3 below, Sweden is divided into different ventilation zones, the associated ventilation factors are in the table beside it.



Ventilation zone	Ventilation factor (F_v)
1-2	1,0
3	1,1
4	1,4
5	1,6

Figure 3. Ventilation zones and associated factor in Sweden (Swedish Transport Administration, 2016)

In step two the exposure units per kg in the area is multiplied with the estimated cost of air pollution per kg. The ASEK-report recommends EUR 61,93 per kg $PM_{2,5}$.

Both the study by Blidberg et al. (2013) and Höjgård and Wilhelmsson (2012) use this calculation (though they use a price per kg $PM_{2,5}$ from an earlier version of the ASEK report) when evaluating the saved cost of emitting less particles. However, Höjgård and Wilhelmsson (2012) argue that the values from ASEK could be associated with uncertainties why they complement their analysis with lower values obtained from HEATCO at EUR 53,5. HEATCO is an EU-funded research project aimed at developing tools and methods as well as approaches to assess costs in the transport sector. They use another way of measuring concentrations and another way of valuing health effects compared to the ASEK-reports. The concentration levels HEATCO use is customized for Continental Europe. Sweden has a more stable climate compared to Continental Europe why the concentration of particles remains in the air longer. Using the HEATCO values in Sweden could therefore lead to an underestimation of particle emissions (Höjgård and Wilhelmsson, 2012).

Another method, used in Brännlund et al. (2010), Bickel et al. (2006) and Tufvesson et al. (2013) is to use three different values to evaluate the socioeconomic value of reduced particulate emissions depending on where the particles are emitted. For rural areas, EUR 42 / kg is assumed, for smaller urban areas EUR 211/kg and for larger urban areas EUR 422/kg. The values are adjusted to assumptions concerning where the emissions emerge. If for instance the production of biogas take place in a rural area and the end use in a larger urban area the emissions from production is valued EUR 42/kg and the emissions from the end use is valued at EUR 422/kg.

Just as for greenhouse emissions, there is a time aspect of the damage cost of particles as well. It makes sense to incorporate an increasing damage cost over time to not underestimate the cost of particle emissions (Nerhagen et al., 2005). Above that, WTP studies are dependent on the respondent disposable income. With time the real income per capita is expected to rise why the real WTP will also rise (Swedish Transport Administration, 2016).

Hence, both increased damage cost and increased WTP is associated with an increased value of reducing particle emissions. However, the difficulty of evaluating this effect in practice is discussed in Nerhagen et al. (2005) with the emphasize that the long-term effects are dependent on many assumptions about the area affected. The study present two scenarios where the one that includes increased damage costs receive a three times larger cost for particle emissions compared to the one that did not on a 20-year calculating time. In the ASEK-report it is recommended to adjust for this increased value by increasing the damage cost by 1,5% annually.

Selected Values

The method for obtaining values used in this study is the two-way method from the ASEK-report. It is used since it allows for more precise calculations as the effect can be adjusted for the scale of population in the studied area. In step one the ventilation zone is assumed to be zone 1, both the Stockholm archipelago and Gotland are in this zone why it seems most reasonable that Åland is as well.

It is difficult to ascertain where emission reduction will have effect and hence what population will be affected from the biogas scenario. An assumption of where the reduction appears is calculated from the municipalities share of Åland´s vehicle stock. The assumption made is that the reduction of particles is evenly spread over all car owners on Åland. Mariehamn has a share of 42% of all cars on Åland, hence 42% of the particle reduction is felt for a population of 11 000. Further, 22% of the car stock are owned by car owners living in municipalities with around 3000 inhabitants, why 22% of the reduction are affecting 3000 inhabitants locally, and so on. Four different populations have been derived. Figure 4 show shares of the car stock on Åland. A table for calculating these shares from car stock and population data are presented in Appendix 2. Figure 5 show the exposure units per kg particles in the associated area, calculated using Equation 4.

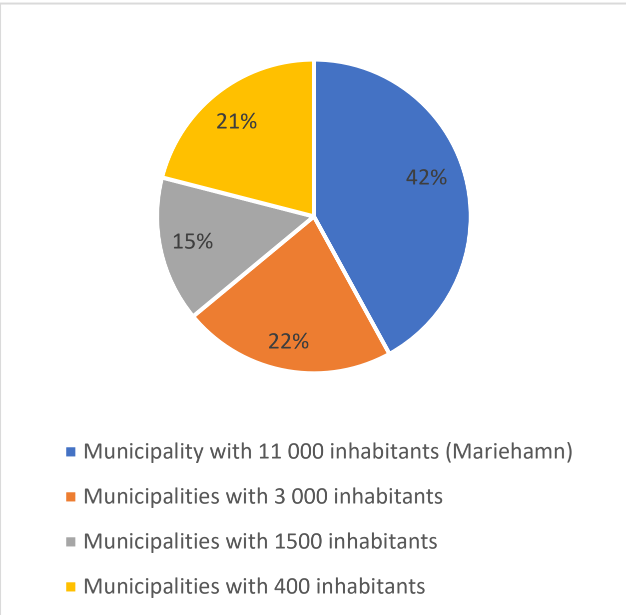


Figure 4. Share of car stock on Åland.
Source: Statistics and Research Åland (2017b)

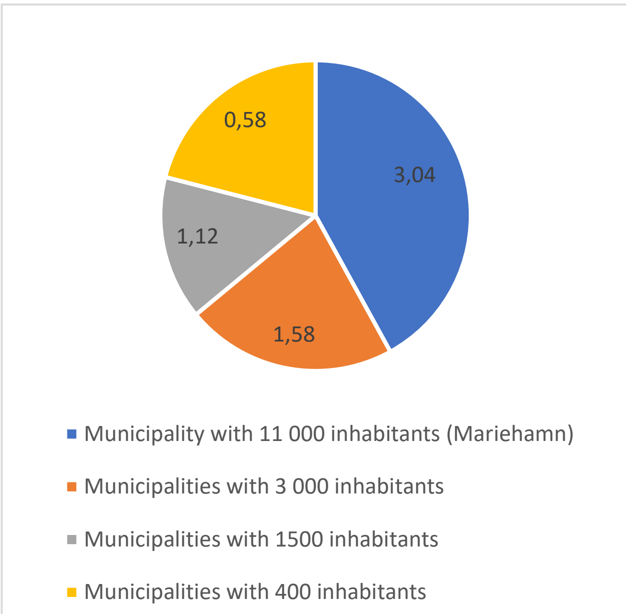


Figure 5. Exposure units per kg particles

Step two is to multiply the exposure units with the cost of particle emission. The cost of emission is set to EUR 61,93 per kg emission as in the ASEK-report. Finally, these costs are multiplied with corresponding reduction share and the total reductions calculated in Table 5. See Table 6 below for price per kg emission in the affected are and the total revenue from reduced particles.

Table 6. Total value of particle emissions

Affected Area	Price per kg emission	Total revenue from reduced emissions
Municipality with 11 000 inhabitants (Mariehamn)	3,04 * 61,93 = 188,27	188,27 * 0,42 * 59,95 = 4740,45
Municipalities with 3 000 inhabitants	1,59 * 61,93 = 98,47	98,47 * 0,22 * 59,95 = 1298,70
Municipalities with 1500 inhabitants	1,12 * 61,93 = 69,36	69,36 * 0,15 * 59,95 = 623,71
Municipalities with 400 inhabitants	0,58 * 61,93 = 35.92	35,92 * 0,21 * 59,95 = 452,21
Total Value from reduced particles		EUR 7115

In the sensitivity analysis, the method presented by ASEK where the price of emissions is increased with 1,5% annually will be presented. For best and worst case scenario numbers from ASEK of EUR 88,46 and from HEATCO of EUR 53,5 will be used.

7.2.3. Reduced environmental impacts from Nitrogen leakage

As described in earlier sections, the rest product from digestion can be used as bio fertilizer and thus replace artificial fertilizer. As the bio fertilizer has a more efficient nitrogen for agriculture this brings a total reduction of nitrogen leakage to the sea from agriculture on Åland.

The valuation of reduced nutrient supply to the ocean differs from the valuation of greenhouse gases and particles. The problem of nutrient supply is a regional problem, as opposed to the greenhouse gas problem, which is a global problem and the problem of particle emissions, that is a local problem (Brännlund et al., 2010). The regional nature that this benefit achieves depends on where the nitrogen leakage occurs. In the following sections the nitrogen leakage will first be quantified by the amount of bio fertilizer that can be produced from this project together with assumptions of how large reduction in nitrogen leakage that correlates with. Following that is the monetization of nitrogen leakage where the societal cost of this reduced leakage is discussed based on the valuation literature as well as defined for this project.

Quantification of nitrogen leakage

The amount that every ton digested substrate can replace mineral fertilizer depend on the share of nitrogen that exists in the original substrate and the content of the mineral fertilizer. To quantify this, a calculation method from Tufvesson and Lantz (2012) is used which is the only study found that measure how much biogas production from fish waste reduce nitrogen leakage. They measure that per every ton digested fish waste the nitrogen reduction to the sea is 0,02 kg. This calculation is reasonable in contrast to the calculations that Brännlund et al. (2010) find for nitrogen reduction of manure (0,027 kg per ton substrate) as well as the calculation used in both Höjgård and Wilhelmsson (2012) and Börjesson and Berglund (2003) where one ton of manure reduce nitrogen by 0,025 kg per ton substrate.

For 2480 ton of fish waste this equals a reduction of 46,9 kg nitrogen per year. To bear in mind is that this reduction is a national average for Sweden why the earlier studies as well as this one will acknowledge that this is a sensitive measure.

Monetization of nitrogen

There are basically two types of studies that can be used to evaluate the supply of nitrogen, one is to evaluate the willingness to pay (WTP) for reduced nitrogen supply in the sea. The other is to evaluate the WTP for improved visibility depth in the sea. Hence, both use a WTP-method were people state their preferences. As mentioned in the earlier section about CBA, these kinds of valuations are associated with bias. This since the studies are hypothetical, meaning that the respondents do not actually have to pay the amount they specify, why they tend to overestimate the willingness to pay.

In the report "Monetary standard values for environmental change" by the Swedish Environmental Protection Agency, (Kinell et al., 2009) have evaluated eight different studies on the value of reduced nitrogen leakage to the sea. The considered studies are a mix of both WTP for nitrogen reduction and WTP for improved visibility. Five are contingent valuation studies and three are travel cost method studies. The value of one kg nitrogen reduction ranges between EUR 1,23 and EUR 23,53 with a standard value of EUR 8,25. To correct for the bias of overestimating preferences Kinell et al. (2009) use a method where the stated willingness to pay is divided by 3 which cause the range to go from EUR 0,41 – EUR 7,84 with a standard value of EUR 2,75. Several studies base their socioeconomic valuation from this calculation, both Brännlund et al. (2010), WSP (2012) and Blidberg et al. (2012) use the values adjusted for overestimation bias while Höjgård and Wilhelmsson (2012) and Tufvesson et al. (2013) use the non-adjusted values.

Selected Values

This study will use the adjusted values by Kinell et al. (2009) to calculate the socioeconomic value of reduced nitrogen leakage, in the base case the standard value of EUR 2,75 per kg leaked nitrogen be used.

Though, since it is difficult to determine where in the range of value presented in the literature the exact valuation lies, because this depends on the geographical location two extreme values will be used for a best-case scenario and worst case scenario of the environmental impact of nitrogen leakage. Best case scenario will be calculated with EUR 23,53 and the worst case scenario will be calculated with EUR 0,41 per kg of reduced nitrogen which corresponds to the highest and the lowest values found in Kinell et al. (2009).

In the base case this implies that the socioeconomic reduction of nitrogen has a value of EUR 129 per year. This value is low in comparison to the value of reduced greenhouse gases. This is a general conclusion found in the literature (See Brännlund et al., 2010; Blidberg et al., 2012; Höjgård and Wilhelmsson, 2012). Though despite this it is an important environmental factor that could be increasingly important if more substrates are included in the biogas digestion causing larger scale production.

7.3. Marketable Benefits

7.3.1. Profit of selling biogas

Pricing on biogas vehicle gas is mainly not due to the cost of the producing biogas, but the price is primarily alternative-cost-based. This means that the price of biogas at the pump is set along the price of the options, especially the price of gasoline and diesel (Vestman et al., 2014). Since 2013 the vehicle gas industry has sold biogas in kg, 1 Nm^3 of vehicle gas thus corresponds to approximately 0.8 kg. The price is fluctuating just as the price of gasoline but a recent price in March 2017 corresponds to EUR 1,9 per kg (The Swedish Gas Association, 2017). When comparing the fossil fuel price with the biogas price it is vital to consider that biogas contains more energy than fossil fuels. The price can be calculated into fossil fuel equivalents by dividing the price by 1,6 (ibid), see Equation 5 for the fossil fuel equivalent price in EUR.

$$Fossil\ fuel\ equivalent = \frac{1,9}{1,6} = 1,19$$

(Equation 5)

The gas price on Åland is estimated to be EUR 1,47 for March 2017 (Global petrol prices, 2017) which make the fossil fuel equivalent price in accordance with the assumption that the biogas price to costumers is about 20% less than fossil alternatives. The revenue to biogas producers are evaluated to be 70% of market price both by Allerborg et al. (2015) and Blidberg et al. (2013) why the revenue per kg biogas is estimated at EUR 1,33 (70% of EUR 1,9), see Table 7 for estimated revenue from selling biogas. It is obviously very difficult to ascertain that this price is a good estimate of the price over time. In the sensitivity analysis, a best case will calculate the price of biogas at EUR 1,6 per kg (increase in expected revenue by 20%) and a worst case will calculate at a price of EUR 1,06 per kg biogas (decrease in expected revenue by 20%).

Table 7. Total revenue of selling biogas

Tons of Fish waste	Produced biogas	Converter Nm^3 to kg	Price for kg biogas	Total revenue in EUR
2480 ton	1 076 320 Nm^3	861 056	EUR 1,33	1 145204

7.3.2. Profit of selling bio fertilizer

After digestion to produce biogas, there remains a rest product consisting of hard biodegradable organic matter, inert inorganic material, nutrients, microbes and water. Due to the high nutritional content, this rest product is used as a fertilizer on farmland and is then called bio fertilizer (Brännlund et al., 2010). It is very common to take advantage of this rest product, in fact 99% of all large-scale biogas factories in Sweden use this rest as fertilizer (Swedish Energy Agency, 2016b).

The possible revenue from this waste product is divided in the literature on the subject. Some studies find that this waste product can give a positive net benefit, Stenberg et al. 2013 find that this benefit is about EUR 1 per ton fertilizer. Vestman et al. (2014) instead find evidence that the biogas producer in most of the studied cases pay farmers to collect the fertilizer at a cost from about EUR 10. For some of the studied factories that cost also include transportation the fertilizer to collect spots.

Also, several studies estimate that the cost of preparing the fertilizer for sale and selling it evens out and calculate on a zero profit from selling the fertilizer. Both Blidberg et al. (2012) and WSP (2012) calculate with this zero profit.

Selected Values

The assumptions in this study is that the fertilizer will be demanded from the agriculture on Åland, but because of uncertainties regarding the costs of preparing and transporting the waste product the profit of selling it is set to zero.

7.3.3. Employment effects

The production and use of biogas can also give rise to different employment effects. Several of the studies in the literature review discuss the socioeconomic revenue of employment factors as an effect of increased biogas production.

Domac et al. (2005) recognize the positive effect of employment when evaluation socioeconomic benefits from an increased bioenergy sector. The study distinguishes effects from three different categories; direct, indirect and induced effects. The direct effect is the labour required to produce biogas and any other labour directly linked to biogas operations. The indirect effects are the spill over effects in the economy that arise due to the increased demand in for example subcontractors. The induced effects are the spill over effects because of increased income in the region, as residents spend to some extent on goods and services produced in the region. The study does not give any exact measures on the size of these employment effect but emphasize that it depends on for example stages in the overall bioenergy system, stage of conversion process and whether the sector is increased in a developing or developed country.

WSP Sweden (2012) has made a study where the employment effect of an increased biogas production in Skåne in Sweden is analysed. They also differentiate the effect in the same way mentioned in Domac et al. (2005). The study estimate that 1,1 – 1,4 full year employments would be added per GWh produced biogas. The number shall constitute the additional employment the biogas production give rise to compared to fossil fuels. Stridsberg (2008) evaluate that increased the increased employment effect is 200 full year employments per TWh, which corresponds to 0,2 employments per GWh. Stridsberg is measuring the employment effect from biogas production from manure and Börjesson (2007), who mentions the values from Stridsberg, emphasize that the employment effect is lower when biogas is produced from waste. Hence, the interval is quite large regarding the employment effects from biogas production.

However, a large interval is not the only issue when evaluating socioeconomic impacts from biogas production. Brännlund et al. (2010) question whether there are any positive net effects from employment at all. The study argues that investing resources in one sector in the economy means, in the long run, that resources need to be taken from other sectors. Therefore, Brännlund et al. (2010) claims that there is strong support that the net effect in the long term is zero. This is an argument that Höjgård and Wilhemsson (2012) agrees to as they conclude that biogas production is not the only employment opportunity open to those occupied, why there are no external societal benefit from that employment per se.

Selected Values

Since the employment effects are so uncertain in general and because a precise valuation requires close studies on the labour market on Åland, this study will not monetize the employment effect. Instead, the sensitivity analysis will include an estimation of the value of the employment effect needed for the biogas scenario to break even.

7.3.4. Additional effects

Apart from the already mentioned impacts for biogas production from fish waste, there are some additional effects that is found among the literature but that will not be evaluated further within this study. Among these are living countryside, an impact that is for example discussed in Anderson et al. (2016) which could be increased as the biogas production occurs in rural areas. However, this is a political goal that is difficult to value (Anderson et al., 2016). Another is reduced noise pollution as biogas cars generally have a lower level of noise pollution. These levels are also very difficult to monetize as it requires information of the affected population per mileage (WSP Sweden, 2012). Yet another positive benefit from increased biogas production is energy independence, which refers to the value of security of energy and not being dependent on importing energy. For further reading see for example de Sampaio Nunes (2002).

8. Chapter Eight – Results

The result section will present the result from the CBA annually and with Net Present Value (NPV) to analyse whether the biogas scenario is socioeconomically profitable in comparison to status quo. The robustness of the NPV is later tested in the Sensitivity Analysis. Table 8 shows an overview of the identified costs and benefits generated from the biogas scenario in nominal values annually.

Table 8. Annual Net benefit, nominal values in EUR

Costs	EUR
Opportunity Cost from Reference scenario	- 467 400
Transportation	- 36 742
Production	- 1 144 882
Environmental Benefits	
Reduction of GHG-gases	277 223
Reduced particles	7 115
Reduced Nitrogen Leakage	129
Marketable Benefits	
Selling biogas	1 145 204
Selling bio fertilizer	0
Employment benefits	0
Annual Net benefit	- 219 353

8.1. Net Present Value

The Net Present Value (NPV) of the study is the sum of costs and benefits over all the years the project is assumed to have societal value. The NPV is converted into present value terms by the discount rate. Thus, the values presented onwards are real values. The interest rate chosen in the base case is 3,5%. Equation 6 below show the NPV of the biogas scenario over 15 years.

$$NPV = \frac{-219\,353}{(1+0{,}035)^1} + \dots + \frac{-219\,353}{(1+0{,}035)^{15}} = -2\,745\,732$$

(Equation 6)

The project showed a negative NPV of EUR 2 745 732. This implies that the project, as calculated with base case evaluations, is not socioeconomically profitable.

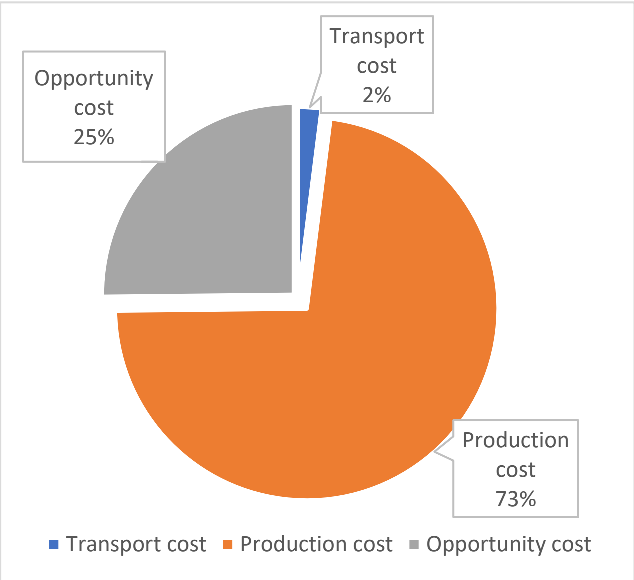


Figure 5. Shares of total costs (real values)

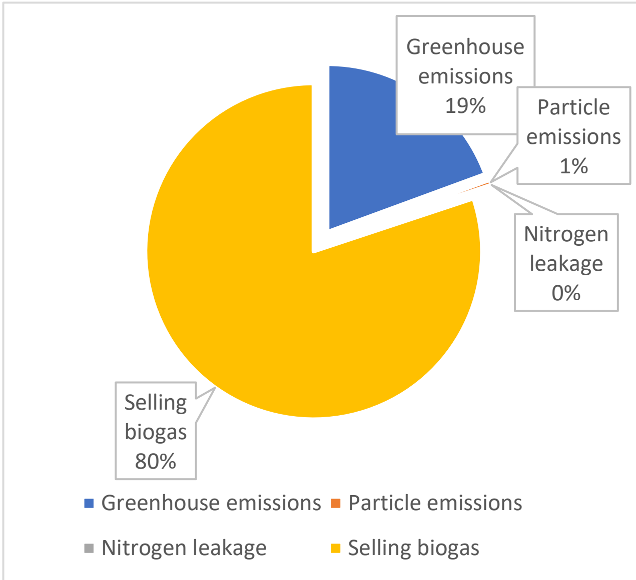


Figure 6. Shares of total benefits (real values)

Figure 5 and 6 show the shares of the total costs and total benefits. As can be seen production cost is the largest cost of the project and the largest benefit is selling biogas. The benefit of nitrogen leakage is so low (0,009% of total benefit) it does not even show in the diagram.

8.2. Sensitivity Analysis

The purpose of the sensitivity analysis is to tests the robustness of the results when varying different assumptions. Only assumptions regarding valuation of the different impacts will be varied in the analysis, hence the quantification measures will not be tested.

8.2.1. Varying discount rate

As mentioned earlier, a sensitive part of a CBA is the chosen discount rate. With low discount rates the values of the future become more valuable today, which can be seen from Table 9 below where the NPV decreases if future worth is valued higher (lower interest rates). The NPV of the different discount rates has considered that the capital cost of production varies with discount rates as well.

Table 9. Varying time frames and discount rates

Discount Rate	NPV
1%	- 6 354 988
3,5% (Base case)	- 5 351 343
5%	- 4 864 940

8.2.2. Varying production cost

A large share of the costs of the biogas scenario is the production cost. For more than one reason it is appropriate to estimate the variation in NPV when the assumption of production cost are alternated. One reason is that the interchangeability between costs used in other studies might not be as applicable on the studied scenario as assumed, why other costs than the one in the base case would be motivating to evaluate. Another reason is to find the breakeven point of production cost (an NPV of zero). It has been discussed earlier in the study that it might be possible that other substrates could be collected and digested in the biogas factory as well. A larger scale production is associated with a lower investment- and productions costs per kWh.

In Figure 7 below the NPV is estimated as function of the price per one produced kWh. The break-even price per kWh is approximately EUR 0,09 per kWh which shows how small the marginals are for profitability.

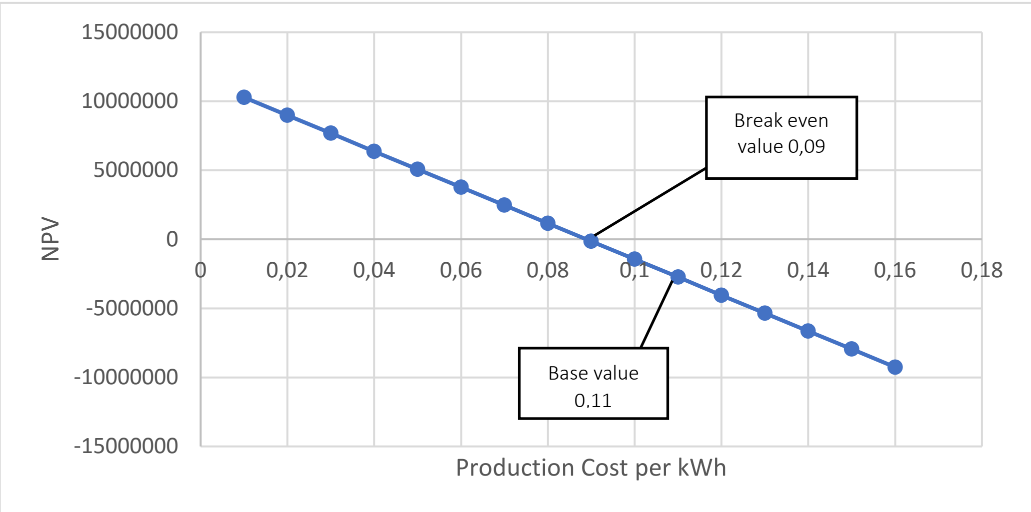


Figure 7. NPV as a function of production cost per kWh

If not only fish waste would be digested in the factory but would be accompanied with the substrates from the study by Allerborg et al. (2015) the total produced biogas would add up to 30 GWh annually, (20 Gwh from Allerborg et al. (2015)[9]. and about 10 GWh from fish waste). If the production scale increases to 30 GWh the cost of investment and operation could be reduced according to Stenberg et al. (2013)[10]. The investment needed is lowered to EUR 0,29 per annual kWh which makes the investment cost only 0,033 per kWh. Operation cost is lowered at EUR 0,02 per kWh. Taking the same cost for distribution from Vestman et al. (2013) at EUR 0,024 per kWh makes the cost per kWh 0,08 which is below the break-even price. Hence, if the total production scale could increase to 30 GWh, the cost of producing biogas from fish waste could decrease so that the biogas scenario would break-even with the reference scenario.

8.2.3. Varying value of greenhouse gases

Another earlier discussed sensitive measurement is the value of greenhouse gases that is estimated with great uncertainty, it is therefore essential to present results with different valuations than the one chosen in the base case. In Figure 8 below the NPV is presented as a function of the EUR value of CO_2e, all other measurements are in accordance with the base case. The figure shows that the price of one kg of CO_2e must be valued above EUR 0,21 for the project to break even. This value is quite far from the one assumed in the base case at EUR 0,12 and even further away from the average found in Tol (2008) at EUR 0,02. However, it is below the highest found value in the literature at EUR 0,35 per kg and below the value presented in the well sited Stern review at EUR 0,25 per kg CO_2e. This demonstrates that is it not completely unlikely that the external benefit for society to reduce emissions lies above the break-even value where the biogas scenario is socioeconomically profitable.

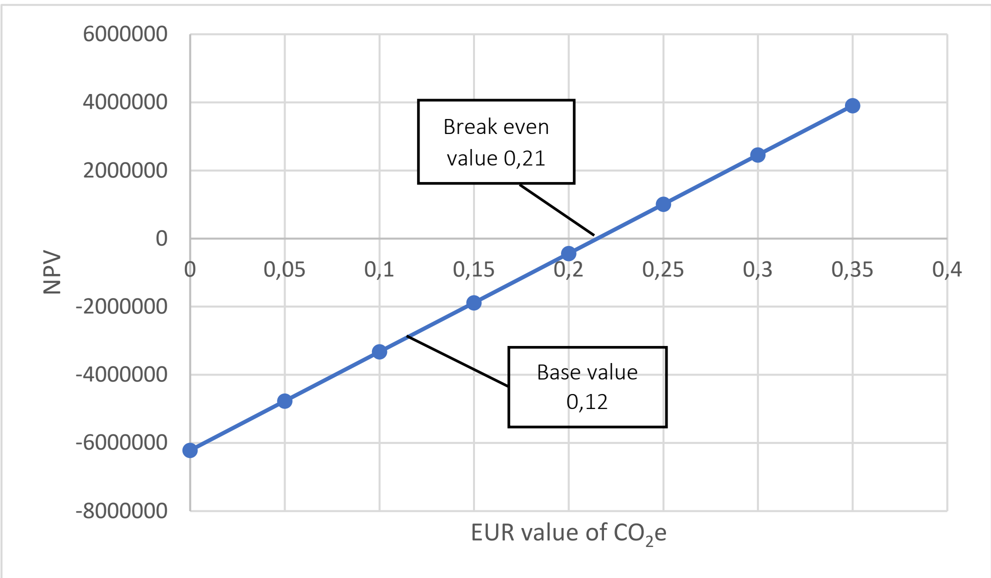


Figure 8. NPV as a function of CO_2e value

[9] A small overlap from that study is however vital to keep in mind as the study include 500-ton waste from the fish industry on Åland.
[10] See Table 3. for different investment and operation cost from Stenberg et al. (2013) at 30 GWh.

8.2.4. Employment break-even analysis

As the employment effects of increased biogas scenario showed to be very difficult to evaluate, estimations will instead be aimed at calculating what the effect need to be in monetary terms for the project to pass the NPV-test. This is done by solving Equation 7 below, where the total value of the employment effect must be larger than the negative NPV of EUR 2 745 732.

$$\sum_{t=15}^{n} \frac{Employment\ effect}{(1+0{,}035)^1} + \cdots + \frac{Employment\ effect}{(1+0{,}035)^{15}} > 2\ 745\ 732$$

(Equation 7)

The answer to the equation is that the employment effect must be greater than EUR 219 353 in nominal value annually. In the literature of employment effects a range was found between 0,2 – 1,4 full employments per produced GWh biogas per year. The production in the biogas scenario amounts to 10,4 GWh which implies that every full year employment must have a socioeconomic value of EUR 15 065 – 105 458 for the project to break even.

8.2.5. Best case vs. Worst case

Table 10 below show the base, best and the worst-case scenarios together with assumptions of whether the environmental benefits are increased over time or not. The opportunity cost from the reference scenario is not altered. The results are in line with the common result of extreme values in CBA, as the variance between the best and the worst case is substantial. The table also shows that even with increased value of 1,5% per year of all the environmental benefits, the NPV is not positive in the base case.

Table 10. NPV at best case and worst case

	No increase in environmental value	With increase in environmental value
Base case values	- 2 745 732	- 2 368 433
Best case values[a]	12 336 672	13 452 002
Worst case values[b]	- 12 860 588	- 12 789 373

[a] Best case (worst case) scenario is valued with transportation cost at EUR 0,07 (0,16) per kilometre transported ton, production cost at EUR 0,07 (0,143) per kWh, CO_2e emissions at EUR 0,35 (0,02) per kg CO_2e, particle emission at EUR 88,46 (53,30) per kg particles, nitrogen leakage at EUR 23,53 (0,41) per kg nitrogen and profit of selling biogas at EUR 1,6 (1,06) per kg biogas.

9. Chapter Nine – Conclusion

The purpose of this study has been to evaluate whether biogas production from fish waste is a more profitable reutilization compared to the use today. To investigate this issue a Cost Benefit Analysis (CBA) has been performed where the cost and benefits form a defined biogas scenario have been identified. The study identified and valued reduced environmental impact from reduced greenhouse gas emission (CO_2e emissions), reduced particle emission and reduced nitrogen leakage to the sea together these environmental benefits stood for 20% of the net benefits. Alongside the environmental benefits, the profit of selling the biogas was identified as an important revenue, and it was found to be the largest revenue with 80% of the net benefits. On the cost side, the largest share was the biogas production which stood for 73% of the net costs. The opportunity cost of the reference scenario was as well found to be a vital cost with 25% of net costs. The transport costs showed to be small in comparison with its net cost share of 2%. All the identified goods were valued with benefits transfer from earlier studies or by market prices.

The result section showed that the base case was not socioeconomically profitable regardless of changed interest rates or increased damage cost of the environmental impacts giving increased benefits for reducing emissions. The general result is that, under the current assumptions and conditions of the biogas scenario, the NPV<0 and in accordance with the common decision rule to perform projects if the NPV >0, and to reject projects if the NPV<0, the project should not be performed. This leads to the conclusion that the answer to the question of research is no, biogas production is not a socioeconomically profitable solution for reutilizing fish waste on Åland.

The socioeconomic value showed to vary considerably depending on the assumptions and methods used for valuing the identified impacts. This was particularly seen in the best/worst case analysis. This insecurity is mirrored in the literature as well as large ranges could be identified for every environmental impact. What also could be concluded from the result is the sensitivity of measurements for the external costs of the project. This could especially be seen in the graph showing NPV as a function of different valuations of a reduced kg of CO_2e. The graph showed that there is uncertainty regarding the societal cost of CO_2e and that the great range in the literature makes it difficult to ascertain if the total NPV for society is above or below zero in the biogas scenario.

The proportion of benefits from both particle emissions and nitrogen leakage have shown to be small in comparison to reduction of greenhouse gases (see Brännlund et al., 2010 for example) but the proportion of benefit coming from particle reduction and reduced nitrogen leakage is exceptionally small in this study. For particle reduction, this is most likely since the affected population on Åland is so much smaller than the populations expected to be affected in the literature. For reduced nitrogen, the benefit is likely low since the substrates from several of the other studies were not reutilized whatsoever in status quo. If the scenario of reference did not process any of the fish waste the benefit of nitrogen leakage would have been greater. An important remark to bear in mind if similar projects are studied where the reference scenario implies disposing the fish waste at sea.

In addition to the presented socioeconomic value of the biogas scenario, the study also identified positive employment effects and possible revenue from selling bio fertilizer. The employment effect is in earlier studies estimated to be between 0,2 – 1,4 full yearly occupations per GWh biogas. As showed in the sensitivity analysis, if the employment effect is valued above EUR 219 353 in nominal terms annually, then the socioeconomic profitability will be superior to the status quo.

Over all, the results are consistent with likely studies in the field, neither Stenberg et al. (2013), nor Blidberg et al. (2013) reached socioeconomic profitability with a comparable production scale. A likely common reason for this negative result is the price of the substrate. When evaluating socioeconomic profitability Tufvesson and Lantz (2012) emphasise that the cost of the substrate is substantial. For both mentioned studies, it is foremost the cost of collecting the substrate that is too costly, for Blidberg et al. (2012) it is the harvest of reed and for Stenberg et al. (2013) it is the cultivation of sea squirts. For this project, it is the opportunity cost that is a major stick in the wheel for profitability. Almost all the fish waste from the fish farms on Åland today are sold at EUR 200 per ton which amounts to a large opportunity cost every year. If the fish waste would simply be tossed after it has been cleared away in production, 25% of the costs would be reduced and the NPV would be above zero. As stated by (Tufvesson and Lantz, 2012), high opportunity costs are one common reason for negative socioeconomic outcomes.

One way to forgo the high cost of the substrate, as was demonstrated in the sensitivity analysis, is to decrease production cost. This could be done by increasing the volume of substrate and therefore also the total biogas produced. A scenario that is not unthinkable on Åland as the earlier study by Allerborg et al (2015) estimated the total biogas potential to 20 GWh. The study by Stenberg et al (2013) also received a positive result when increasing production scale to a bit over 30 GWh annually.

For future research, it is recommended to include effects that were identified, but not monetized within this study to get a more precise estimated value then could be estimated in this study. Also, it would be valuable to increase assumed scenarios to include the estimated effect if biogas was not only assumed to be replaced in passenger cars on gasoline but also busses and heavier traffic. Lastly, an investigation of what policy instrument would be best suitable to support biogas production in the case of higher production scale would be a natural continuation in the field of study.

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Appendix

Appendix 1. Transport cost

Fishfarm	Fishwaste per year	Distance to biogas plant	Kilometre transported ton
1	280	117	32 760
2	70	16	1 120
3	120	53	6 360
4	480	44	21 120
5	1 000	44	44 000
6	140	37	5 180
7	450	12	5 400
Total	2 540	323	115 940

Source: Added value from fish waste (Åland University of Applied Sciences, 2017)

Appendix 2. Car share and associated population

Municipality	Number of cars	Population	Share of car stock
Brändö	416	471	1%
Eckerö	1 108	928	4%
Finström	2 785	2 594	10%
Föglö	568	561	2%
Geta	527	499	2%
Hammarland	1 636	1 508	6%
Jomala	3 380	4 757	12%
Kumlinge	292	308	1%
Kökar	197	246	1%
Lemland	1 533	2 012	5%
Lumparland	377	385	1%
Mariehamn	11 896	11 565	42%
Saltvik	1 733	1 839	6%
Sottunga	96	96	0%
Sund	1 153	1 006	4%
Vårdö	424	439	1%
Other	444		2%

Share of car stock	Associated population in home municipality
42%	11 000
22%	3 000
15%	1 500
21%	400

Source: Statistics and Research Åland, 2017b