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# Impact Assessment Modelling for the Ocean Economy: A Review of Developments

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# Impact Assessment Modelling for the Ocean Economy: A Review of Developments

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# 1. INTRODUCTION

The achievement of a sustainable ocean economy requires the collection of relevant sectoral data the development of suitable indicators and the provision of appropriate analysis so as to aid policymaking. The importance of the marine and ocean economy can be seen in the extent of the world's oceans, the proportion of the world's population living in coastal areas and the aims of Sustainable Development Goal 14 which looks to "conserve and sustainably use the oceans, seas and marine resources for sustainable development" (UN, 2015). The ocean economy is also an important driver of the European Green Deal, with an emphasis on ensuring sustainability and creating new green jobs and businesses across the sector (European Commission, 2021).

As an EU economic sector with a turnover of €750 billion and with almost five million people employed in 2018, the ocean economy has been identified as a driver of European growth through the development of new competences and activities that enable a sustainable development of ocean resources (European Commission, 2020). As a part of the bio-economy, it also has a large potential in terms of its contribution to a green recovery. Coastal regions, home to over 40% of European citizens, also have much to gain from the European Green Deal, both economically and environmentally.

With increasingly complex objectives, decision makers require more sophisticated analytical tools with which to design effective policies and implement high-level strategy. However, scientific know-how alone is not sufficient to derive good policy. Bennet (2019) argues that "understanding the human dimensions of the world's peopled seas and coasts is fundamental to evidence-based decision-making across marine policy realms, including marine conservation, marine spatial planning, fisheries management, the blue economy and climate adaptation". There remain serious difficulties in terms of quantifying marine economic and social impacts, thus making it more difficult to make strategic decisions (Foley et al., 2014; Surís-Regueiro et al., 2021). In a review of social scientists, McKinley et al., (2020) highlight a gap in the availability of data for policymaking in the marine space. An extension of this concern, given inadequate data availability, is the limited availability of policy impact assessment modelling.

This paper contributes to the literature by defining a research agenda for policy impact assessment in the marine and ocean economy. Previous work, such as Kerr et al. (2014), which focuses specifically on marine renewable energy or Burbridge et al. (2001), which focuses on the granular detail of impact assessment in aquaculture, focused on specific aspects of the ocean economy. This paper provides a multi-dimensional plan to analyse both individual ocean economy industries and the ocean economy as a whole.

The International Association for Impact Assessment defines impact assessment as a structured process for considering the implications, for people and their environment, of proposed actions while there is still an opportunity to modify (or even, if appropriate, abandon) the proposals (IAIA, 2021). Impact assessment involves the identification and characterisation of the most likely impacts of proposed actions (impact prediction/forecasting), and an assessment of the social significance of those impacts (impact evaluation). The primary goal of the modelling framework in this paper relates to the objective of providing information for decision making through the analysis of the biophysical, social, economic, and institutional consequences of proposed actions.

There are a range of social science multi-disciplinary frameworks for policy assessment including PEST (Political, Economic, Social, Technological) (Sammut-Bonnici and Galea, 2015) and PESTEL (Political, Economic, Social, Technological, Environmental, and Legal) (Yüksel, 2012). In impact assessment models, technology, law, and politics tend to be treated as exogenous although exceptions exist such as technology adoption analysis (Hyvättinen and Hildén, 2004) or where the output of models feeds into political decision-making that may lead to legal changes. Therefore, in considering the use and development of impact assessment models for policy development, we focus on a subset of four dimensions (with the addition of place), Economic, Social, Spatial and Environmental (ESSE). The modelling framework is broken down as:

- Economic – assess the value chain impact of marine industry policy changes using the Bio-Economy Input-Output Model (BIO)
- Social (and Health) – assess the social and health impacts of policy changes, using the Simulation Model of the Irish Local Economy (SMILE) model
- Spatial – assess the spatial impact on rural coastal communities using SMILE
- Environmental – assess the change in the carbon footprint of marine industries using Life-Cycle Assessment (BIO-LCA).

The paper is laid out as follows. First, there is a literature review of the use of economic, social, spatial, and environmental impact assessment models in the marine and ocean sectors. The methodology section details the impact assessment models used in this paper. The results segment provides a review of analyses that have used the ESSE impact assessment framework. Finally, we discuss issues related to the ESSE framework and the future development of the impact assessment tool.

## 2. LITERATURE REVIEW

*Economic*

A range of economic policy models have been developed over time to undertake economic impact assessment so as to assist policy planning. The dominant analytical framework has been the input-output model, which can track material flows between sectors. Although this methodology has been used to explore impacts across many marine sectors, the availability of quality data has led to numerous assessments of the fisheries sector in particular (Hoagland et al., 2005; Kirkley, 2009; Lee and Yoo, 2014; Grealis et al., 2017; Rizal et al., 2019). In addition to commercial fishing, input-output models have been used also to consider the impact of recreational fishing (Storey and Allen, 1993**;** Steinback, 1999; García-de-la-Fuente et al., 2020). The input-output framework has also been used by

Morrissey and O'Donoghue (2013a) to consider clusters in marine transport and by Morrissey and Cummins (2016) to examine marine-related energy and recreation clusters.

Policy analysis and planning has moved from silo-based approaches focused on for example individual sectors like fishing (Andrews and Rossi, 1986), transport (Goss, 1967) and energy (Frair and Devine, 1975) to more integrated cross-sectoral approaches (Norton and Hynes, 2018). Improved data collection on the wider marine economy (Foley et al., 2014; Morrissey 2014; Wang, 2016; Vega and Hynes, 2017; Tsakiridis et al., 2019) have enabled input-output frameworks to incorporate wider sectoral dimensions (Kwak et al., 2005; Zhao and Wang 2008; Morrissey and O'Donoghue, 2013b; Wang and Wang, 2019; Zheng and Tian, 2021). Additionally, Surís-Regueiro et al. (2021) have utilised input-output analysis in a comparative setting of marine spatial planning in three countries.

One of the limitations of input-output modelling is that it does not incorporate behavioural change, unlike more complex models such as Computable General Equilibrium (CGE) modelling. This can lead to an overstatement by input-output modelling of the relative impact of a policy intervention compared to CGE models, as shown by Allan et al. (2014) in their analysis of marine energy policy in Scotland.

*Social*

There is a relatively substantial literature on the ex-post analysis of the social or distributional impact of the marine sector or its components but there is relatively limited ex-ante impact assessment modelling of these issues. Pomeroy et al. (2007), Pike et al. (2010) and Voyer et al. (2012) argue that a focus on economic consequences often dominates impact assessment, with social impact assessment often underrepresented in marine planning. Ex-post attitudinal surveys on perceptions are often undertaken to gauge public opinion, but after the fact rather than ex-ante so as to aid planning.

One of the most common types of ex-ante analysis is in the use of survey tools to consider the social acceptance of marine developments such as aquaculture (Whitmarsh and Palmieri, 2009). Discrete choice experiments are undertaken to assess the willingness to pay for public good investments and interventions (Rogers, 2013; Jobstvogt et al., 2014; van Osch et al., 2017; Hynes et al., 2013, 2020; O'Connor et al., 2020). Elsewhere, Hatcher et al., (2000) utilise a survey to look at economic, social, and behavioural determinants of regulatory compliance with fishery regulation. Holland et al., (2013) look specifically at understanding the determinants of social capital in groundfish harvest cooperatives in the north-eastern United States. One of the limitations of survey-based instruments is the difficulty in extrapolating from the specific situation considered in the survey to make wider generalisation beyond what is addressed by the survey analysis. Rashid et al., (2016) address this issue by utilising a microsimulation approach to extrapolate from survey data to evaluate the impact of changes in the aquaculture sector on poverty amongst fishermen in Bangladesh.

Analysis of impacts at the micro or individual level is relatively limited in the field, with most such analyses focusing on fishing. Other social dimensions such as age and gender (De la Torre-Castro et al., 2017) are relatively underrepresented in distributional analysis, with the impact on the income distribution more commonplace. Davis and Thiessen (1986) describe the income distribution of Canadian fishermen, while Wamukota et al. (2014) look at the position of local fishermen in Kenya in the income distribution and consider the impact of market integration. Weigel et al. (2015) utilise a multi-faceted micro impact assessment to examine the impact on fishermen of implementing a marine protected area. In general, these analyses are ex-post rather than assessing impacts in an ex-ante fashion.

One field where ex-ante micro-level marine sector analysis occurs more frequently is transport microsimulation. Dougherty (2010), Samimi et al., (2010), Chen and Yang (2014), Fleming et al. (2013), and Grubisic et al. (2020) have used a microsimulation model of land-based queues and transport strategies at marine transport terminals. Goerlandt and Kujala (2010, 2011) and Rong et al. (2015) look at queues and flows of ships outside ports, while Hasegawa (1990) uses this methodology to consider harbour design. Microsimulation models are a useful tool for scenario planning, having also been used for disaster planning. Alam et al. (2018, 2019) have employed these frameworks to look at transport planning in the case of evacuation following a major flood event in a coastal city.

*Spatial*

The role of the marine economy, particularly in peripheral coastal areas, has seen the development of models with a spatial dimension. Some input-output models allow for analyses to be downscaled to single regions (Garza-Gil et al., 2017) whilst other downscale using methods such as location quotients (Morrissey, 2015). In another microsimulation sub-field, spatial microsimulation models have started to be used in spatial impact assessment at the household level. Curtis et al. (2017) utilise a spatial microsimulation model, together with survey data, to quantify the impact of recreational fishing on a remote coastal economy. Morrissey et al. (2014) develop a spatial microsimulation model of the local economy to model the distribution of marine income by sector at a spatial scale in Ireland. Farrell et al. (2020) utilise the framework to assess the spatial impact of a marine renewable energy investment while Hynes et al. (2021) also employ the approach to model the regional employment effects of the Covid-19 pandemic across a range of marine industries.

*Environmental*

Given the increasing interest in environmental issues, input-output analyses have been extended to incorporate environmental impacts (Zheng and Gao, 2015), with additional arguments being made for the incorporation of social dimensions into ecosystem service provision (Martino et al., 2019).

Lin and Nakamura (2019) consider an indirect marine environmental issue in relation to the generation of plastic waste resulting in micro-plastics in the oceans. In relation to marine natural resources, Heen (1989) use a bio-economic model to consider the impact of different harvesting regimes, while Tsakiridis et al., (2020) use an input-output framework to compare the carbon footprint of sea- and land-based protein sources. Elsewhere, Bagoulla and Guillotreau (2020) utilise an input-output model to consider the emissions of the maritime transport sector. Huang et al., (2015) incorporate environmental impact parameters in relation to various pollution sources including air pollution, water consumption and water pollution in an integrated marine spatial planning tool to consider developments in tourism and transportation.

A number of papers have also incorporated multiple dimensions. Wang et al. (2016) link a regional social accounting matrix (SAM) model to an ecological model to incorporate economic, environmental, and spatial dimensions of impact assessment for fisheries. Jin et al. (2003) link an ecosystem matrix with resource multipliers to an input-output model. The authors use this model to illustrate the effects of incorporating habitat destruction and ecosystem structure on resource multipliers when simulating the economic impacts of changes in primary production on final demands for fishery products. Combining spatial, micro, and environmental dimensions, Hynes and O'Donoghue (2020) use a spatial microsimulation framework to incorporate heterogeneous preferences of different population groups on the willingness to pay for water quality improvement. Samuel-Ojo et al. (2015) also use a spatial microsimulation model to simulate marine habitat changes, albeit focusing on the bio-physical dimension of the implementation of a marine protected area strategy rather than modelling the human dimension incorporated in other analyses here.

## 3. METHODOLOGY

The increasing need to develop national and international frameworks to address complex policy issues is a key driver of interest in impact assessment. Some of the requirements from the perspective of marine policy are:

- Breadth - a focus on multi-sectoral aspects of the ocean economy, rather than, for example, fishing and seafood processing or transport;
- Depth - at the same time, there is a need for more in-depth analysis of specific sectors, requiring single sector detail;
- Dimensionality - the impact assessment framework must be able to handle multiple dimensions such as economic, social, spatial, and environmental;
- Systemic - it needs to be able to encompass a system-wide perspective such as the value chain or innovation system.

In this section we shall describe a methodology involving input-output modelling and microsimulation that covers many of these perspectives.

### 3.1 Input-Output Models

National level economic impact analysis of marine and bio-economy sectors in Ireland is undertaken using the Bio-Economy input-output (BIO) model (Grealis and O'Donoghue, 2015). The BIO model has been developed as part of an incremental research programme to disaggregate the national statistical institute's input-output tables, starting initially with the IMAGE model that disaggregated the food sector into sub-components (O'Toole and Matthews, 2002). Later versions updated and streamlined the disaggregation process (Miller et al., 2014) and disaggregated the sea food sector (Vega et al., 2014). Updating the BIO model using 2010 data, Grealis et al. (2015) further disaggregated the model to incorporate the broader marine economy.

Extending the economic impact assessment focused input-output model, O'Donoghue et al. (2019) incorporated life-cycle greenhouse gas emissions for the land-based sectors in BIO-LCA, while Tsakiridis et al. (2020) extended the framework to incorporate GHG emissions for sea fisheries (See Figure 2).

The framework has been used to undertake economic impact assessments in relation to the output and employment impact of national strategies for the Food Harvest 2020 strategy (Miller et al., 2014; Vega et al., 2014), the Food Wise 2025 strategy (Grealis and O'Donoghue, 2015; Grealis et al., 2017), the Harnessing Our Ocean Wealth strategy (Grealis et al., 2015) and discussions are currently underway in relation to its use in the assessment of the Irish Agri-Food 2030 strategy and a possibly updated Irish integrated marine plan.

The BIO model takes the national supply and use tables that are typically updated every five years to disaggregate:

- Primary food production;
- Food processing;
- The marine sectors (see Table 1).

The model disaggregates these sectors using data from the Teagasc National Farm Survey, the CSO Census of Industrial Production and data collected in preparation of the Irish Ocean Economy report (Tsakiridis et al., 2019).

The CSO supplies a number of data sets that provide information on turnover, gross value added (GVA) and employment for all production sectors in the economy. This data is collected across a number of censuses and surveys. In many cases, the data collected is largely concerned with production activity: net output/turnover, input, value added, and employment. However, there are a few data sets which also provide information on the nature and volume of each industry's intermediate consumption, i.e. the composition of their inputs which is required in order to construct an input-output table. The CSO census and surveys which provide data on Ireland's marine sectors include the Census of Industrial Production (CIP) 2007-2012, the Annual Services Inquiry (ASI) 2007-2012, the Building and Construction Inquiry (BCI) 2007-2012 and Intrastat 2007-2012. The data relating to marine activity from these censuses and surveys is provided at the NACE four-digit level. The NACE code system is a pan-European classification system that groups enterprises according to their business activities by assigning a unique 2-, 3- and 4-digit code to each industry. Marine related NACE codes can be fully or partially marine activities. In the latter case, proxies are used to identify the percentage attributable to the marine sector in the NACE codes (see Tsakiridis et al. (2019) for more details).

Table 1 describes the bio-economy input-output marine sectors, their NACE codes, their sub-sectors, where applicable and their primary data sources.

Table 1. Bio-Economy Input Output Marine Sectors

| Sector | NACE Codes | Sub-Sector | Primary Data Sources |
|---|---|---|---|
| Sea Fishing | 03.1 | Sea Fishing | BIM |
| Aquaculture | 03.2 | Aquaculture | BIM/SEMRU |
| Oil and Gas | 06.1, 8.12 and 09.9 | Oil and Gas | CIP |
| Seafood Processing | 10.2 | Seafood Processing | CIP |
| Marine Manufacturing Engineering and Construction | 30.1 | Marine Transport Equipment | CIP |
| | 33.15 | Marine Repair/Installation | CIP |
| | 42.91 | Marine Construction | BCI |
| | 71 | Marine Engineering | SEMRU |
| Marine Retail Trade | 47.23 | Marine Retail Trade | ASI/SEMRU |
| Marine Shipping and Transport | 50.1 and 50.2 | Marine Water Transport Services | ASI |
| | 52 | Marine Warehousing | ASI |
| | 77.34 | Marine Rental and Leasing Services | ASI |
| Marine Tourism | 55-56,79 | Marine Tourism | SEMRU/Fáilte Ireland |

It is assumed that each NACE disaggregated marine sector only produces products that can be classified according to its matching Classification of Products by Activity (CPA) and that no other sector produces those products. This means that each marine sector can be disaggregated from its parent sector directly from the values displayed in the original input-output table without the need to reconstruct the input-output table from a newly disaggregated supply table. In the creation of the disaggregated input-output table, the aggregate figures from the original published input-output table for 2010 from the CSO are assumed to be correct with all balancing adjustments made with respect to preserving these values.

With respect to product taxes and subsidies, in almost all instances reliable information on product taxes and subsidies was not available for the disaggregated marine sectors. The nominal values for the "Product Taxes less Subsidies" row were calculated pro-rata on the basis of the ratio of total output from the sub-sector over the sector or sectors (the latter in the case of "Marine Tourism") from which they are disaggregated. Similarly, where data was unavailable on the individual components of GVA, estimates are based on the ratio of total output from the sub-sector over the sector from which it was disaggregated multiplied by the GVA reported in the Irish Ocean Economy Report (Tsakiridis et al., 2019).

Where it is logically assumed that output from a sector that has a disaggregated marine component flows to another sector that has a disaggregated marine component, the table will reflect that inter-marine sectoral product flow. For example, for the fishing sector it is assumed that output flows from "Repair and Installation of Machinery" (NACE 33), "Construction" (NACE 42) and "Rental and Leasing" (NACE 77) will come from the newly disaggregated marine sector element of those sectors.

While a number of different methods were investigated to aid in the balancing of input-output tables, the decision was taken to balance the disaggregated table manually. While a number of balancing methods such as Cross-Entropy and GRAS were considered, some unexpected results and, in some case, perverse outcomes were observed. All values across the newly disaggregated rows and columns require individual scrutiny and must be deemed credible in the context of the original input-output table and in the face of expert sectoral knowledge. Pragmatic balancing decisions have been made where significant imbalances were detected, particularly the destination of product outputs where very little information is available. Any remaining nominal imbalances have been balanced though "Final Demand".

Figure 1 describes the main purposes of the BIO model; either to simulate the impact of a government strategy or to simulate the multiplier from investment in a novel technology. The Leontief Inverse Matrix for the model is defined as normally as follows:

$$x = (I - A)^{-1}.d \qquad (1)$$

Where $d$ is final demand, $x$ is final input, $I$ is the identity matrix and $A$ is the input-output matrix, defined elements $a_{ij}$, representing the amount of input $i$ $(x_{ij})$ required per unit of output $j$ $(x_j)$. The sum of the columns generates the multiplier. The model allows for either Type I (direct and indirect) or Type II (direct, indirect, and induced) multipliers to be calculated.

A strategic analysis, where a series of targets or objectives defined as $O$ generates an output multiplier $x^*$:

$$x^* = O.(I - A)^{-1}.d \qquad (2)$$

Applying employment coefficients $E$, either average $E_A$ from the ratio of workers per unit of output or marginal $E_M$, derived from a statistical model of marginal employment for each sector as a function of marginal output, we can simulate the employment multiplier of the strategy:

$$e^* = E.O.(I - A)^{-1}.d \qquad (3)$$

In order to simulate greenhouse gas emissions, environmental burden coefficients $r$ (the ratio of environmental burdens to output for each sector) are applied to find a vector of total environmental burdens associated with final demand $(g^*)$, denoted *e* (Kitzes, 2013). This results in a hybrid input-output life-cycle assessment model (a combination of EE-IO and Process LCA) known as BIO-LCA to analyse emissions embedded in the value chain in Ireland similar to Munksgaard (2001).

$$g^* = r.O.(I - A)^{-1}.d \qquad (4)$$

In the BIO-LCA, the environmental burden coefficients, expressed as carbon dioxide equivalents ($CO_2$eq.) per million euros of output, include emissions from energy consumption as well as process emissions (e.g. animal and soil emissions from agriculture). The *e* matrix captures both direct and indirect (or total) emissions that originate from sales to final consumers (Kitzes, 2013). Direct emissions arise as a result of activities directly related to production $(rI)$. Indirect emissions are associated with direct and indirect suppliers and are the difference between direct and total emissions.

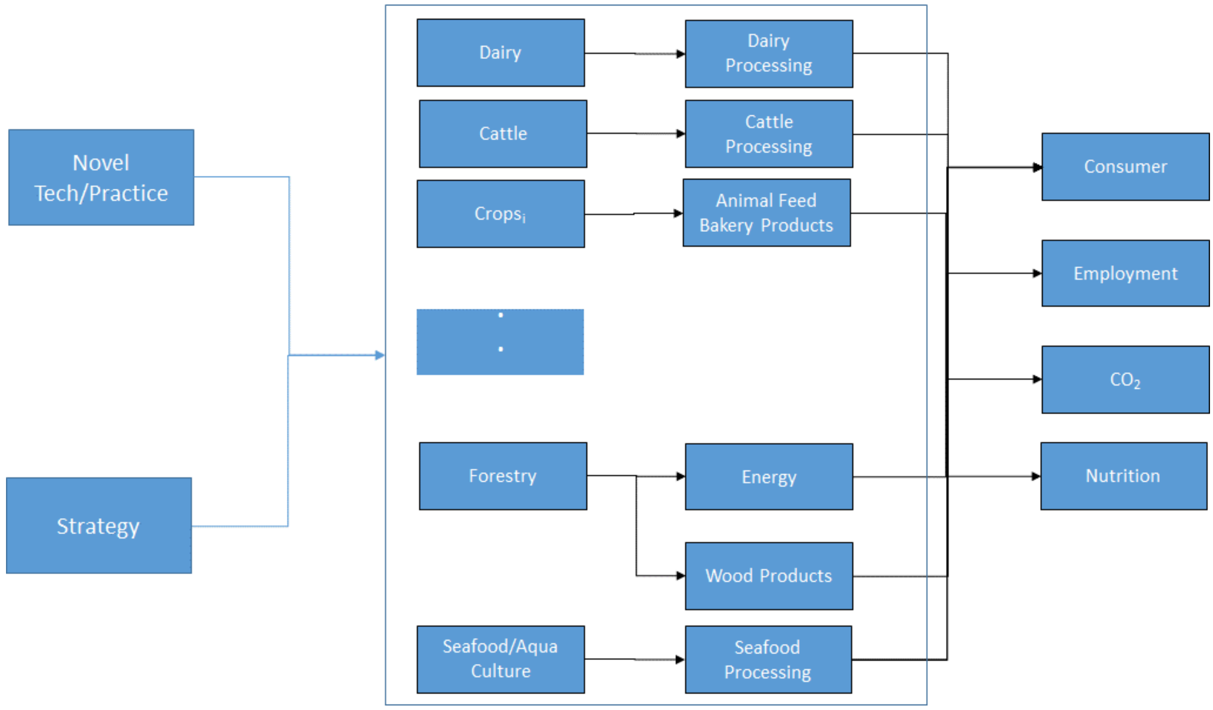


Figure 1. The Bio-Economy Input-Output Model Framework

The other type of analysis used by the modelling framework is to simulate the development of new sectors deriving from novel technologies. This typically involves developing a value chain map like that described in Figure 2, which describes the main connections in the value chain, outlining the structure of the value chain with flows, potential flows, and sectoral actors. New technologies have future and unpredictable impacts and so one often does not have sufficient historical information to assess the potential impact. The modelling framework considers five different dimensions:

Engineering cost and efficiency of the model

The impact of risk and volatility

The economic impact upon the user and developer of the technology via learning

The impact upon the national economy

The environmental impact of the technology

The cost structure is calculated using engineering or pilot plant data to produce a new input-output matrix $A'$ and consequential Inverse Leotief matrix $(I - A')^{-1}$, enabling us to perform economic and environmental impact analysis as above:

$$x'^* = O.(I - A')^{-1}.d \qquad (5)$$

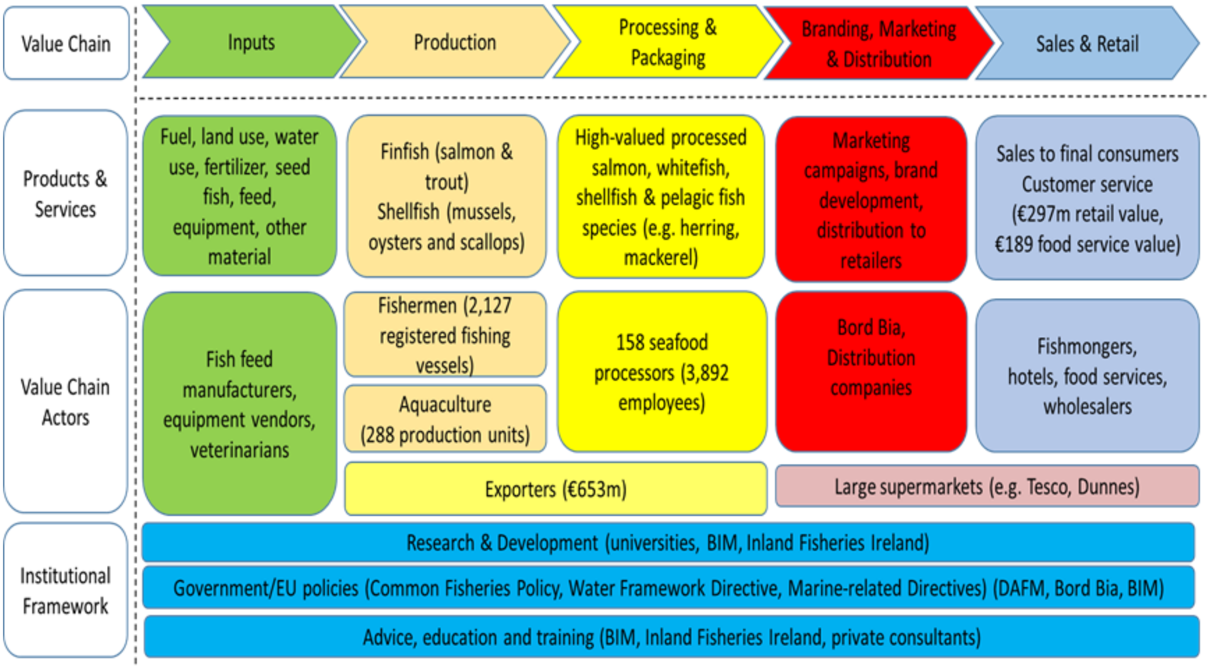


Figure 2. Value Chain Map of the Aquaculture Sector

Source: Tsakiridis et al., (2020)

### 3.2 Spatial Microsimulation Modelling

In order to assess social and spatial dimensions, we utilise a model that combines both micro distributional characteristics and spatial characteristics. Although a regional input-output model has been developed (O'Donoghue, 2021), lack of data availability has meant that the regional model uses aggregated sectors, so disaggregated marine sectors at this spatial level do not exist. Additionally, while Hynes and Farrelly (2012) have utilised small area population census data to characterise the coastal economy, census data in Ireland does not contain income, nor are sectors disaggregated into marine sectors. In order to undertake impact assessments of changing policy or technology, our modelling framework supplements existing aggregate-level economic and environmental assessment by providing a spatially explicit distributional analysis of both the costs and benefits of change to be imposed on households. This is carried out by applying a novel methodology to the field of impact assessment: spatial microsimulation (O'Donoghue, et al., 2014; O'Donoghue, 2014).

Spatial microsimulation has previously been used to estimate the distributional effect of public policy and economic change (O'Donoghue et al., 2012). Although well established in the area of redistributive policy analysis, spatial microsimulation is still an emerging field for assessing regional employment changes. To date, studies employed have either been of a demonstrative nature (Ballas et al., 2006) or have focused on population dynamics (Rephann et al., 2005).

Figure 3 describes the broad structure of the Simulation Model of the Irish Local Economy (SMILE). The fundamental analytical driver of much of the spatial microsimulation literature is to link together data with different attributes to undertake policy analysis with both spatial and social or distributional implications. Many datasets contain one or other of these components, but it is rare to have data with both spatial and social dimensions. Spatial microsimulation models use data enhancement techniques like simulated annealing or quota sample matching (O'Donoghue et al., 2014) to produce coherent spatial and social data.

SMILE uses quota sample matching to sample data from a relevant micro dataset to be consistent with small area spatial data. While there is a farm-level model (O'Donoghue, 2017), the marine disaggregated analyses focus on the household unit of analysis, where households are sampled from the Eurostat Survey of Income and Living Conditions (SILC) that contains income, demographic, and labour market data to be consistent with the CSO Census of Population small area statistics. Additional data on expenditure is statistically matched into this dataset for consumption analyses. One of the features of the Census of Population is the availability of detailed origin and destination data at a micro level by industry, which allows the researcher to incorporate both place of residence and place of work, giving the commuting footprint of different locations. The model is representative at the level of the electoral division (ED). As in the case of the input-output framework, the impact of novel technologies or strategy goals can be simulated, modelling the impact on income, place, and carbon emissions.

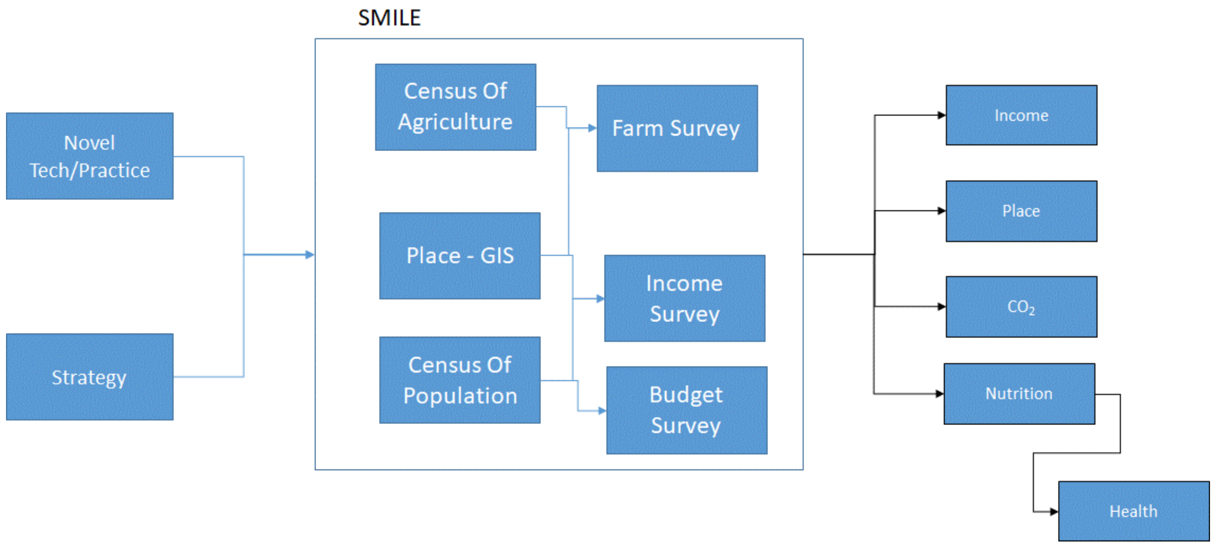


Figure 3. Spatial and Social Impact

As in other analyses of the marine sector, a specific challenge is a lack of availability of appropriate data. SMILE contains aggregated sectors which combine marine and non-marine sectors. In order to disaggregate the marine sectors in SMILE, we utilise two marine datasets collected for the development of the Irish Ocean Economy Report (Tsakiridis et al., 2019).

We utilise data at the county level in relation to marine employment, disaggregated into 12 marine sub-sectors and ED information in relation to the location of marine businesses classified by these sub-sectors.

Utilising place of work data, we can identify the number of workers in relevant sectors across 3,400 electoral divisions. Combining marine employment and business location data using iterative proportional fitting, we derive calibration totals of marine sub-sectors and then randomly allocate workers into these sub-sectors from the aggregate sectors working in those locations. Using the Origin-Destination commuting data, we connect the worker back to their household. Elsewhere in this special issue, Hynes et al. (2021) demonstrate the use of SMILE to model the regional employment effects of the Covid-19 pandemic across a range of marine industries.

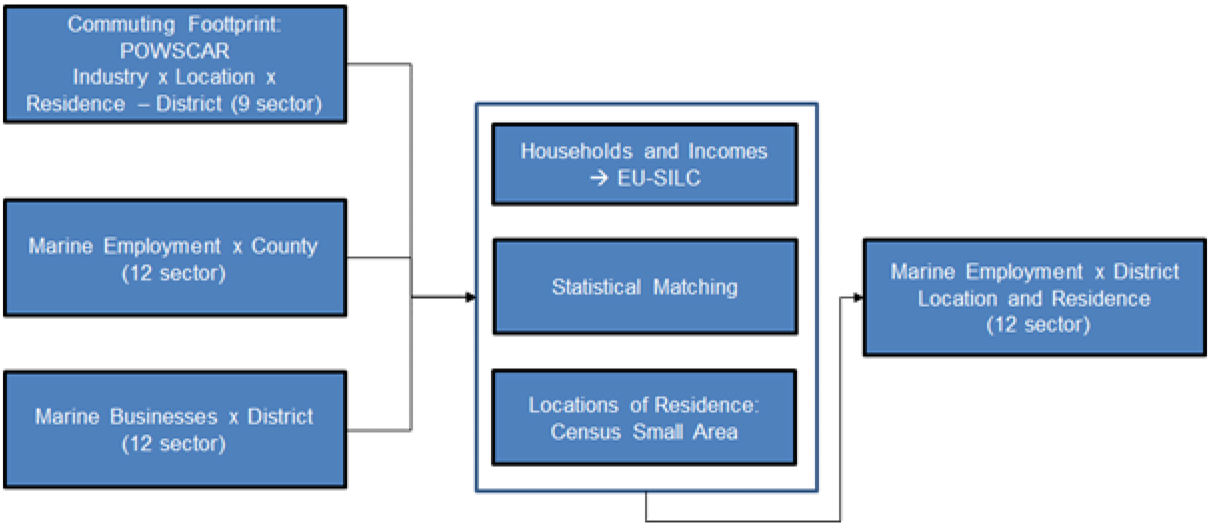


Figure 4. Description of the Marine Disaggregation in SMILE

## 4. RESULTS

In this section a number of analyses using the ESSE impact assessment framework are described, highlighting its functionality. Two analytical examples are considered: an environmental, life-cycle assessment comparison of different land- and sea-based food products and an economic, social, and spatial impact of a novel technology, wave energy installation.

The different dimensions considered across the two case studies are as follows:

- Food life-cycle assessment
  - Direct and indirect multipliers were calculated for a range of land- and sea-based food value chains.
  - An environmental analysis was undertaken to assess the net impact on greenhouse gas emissions of each of these value chains.
- Impact from the installation of wave energy devices

- Scientists were engaged to develop an engineering model to assess the private cost-benefits to the developer.
- Using stochastic assumptions about the variability of input and output components, a risk assessment was undertaken.
- The engineering parameters were incorporated into an input-output framework to assess the impact on the wider economy. At the core of the framework is an input-output model of the Bio-Economy Input-Output (BIO) model.
- Spatializing these inputs and outputs, the spatial development of marine energy devices was simulated within the Simulation Model of the Irish Local Economy (SMILE) to assess the regional impact.

### 4.1 Input-Output Modelling: Extending Value Chain Impact to Incorporate Carbon Footprint

Demonstrating the environmental dimension of the impact assessment, we report on an analysis of the carbon footprint of different protein rich sea- and land-based foods. The global food sector is currently responsible for approximately 30% of global energy consumption and more than 20% of the global GHG emissions (FAO, 2011, as cited in Ytrestøyl et al., 2015). Meat and dairy products account for approximately 18% of these emissions. Substantial increases in food productivity need to be achieved to meet increasing demand for food, while minimizing producers' cost and environmental externalities, such as GHG emissions (e.g. carbon dioxide, methane, and nitrous oxides), ocean acidification and inland fresh water pollution by nitrates, phosphates, and pesticides.

Nijdam et al. (2012) analysed 52 LCA studies of animal and vegetal sources of protein focusing on land requirement and carbon footprints, finding that differences in production systems mainly drive differences between product footprints. Pork and poultry show more homogeneity than beef and seafood. From farm to fork, animal feed production and animal husbandry are the most important contributors to the environmental impacts.

Under the Effort Sharing Decision (ESD) agreed in 2009, each EU member state has been assigned a GHG emissions target for 2020, which represents a percentage change relative to the associated emissions level in 2005. The targeted reduction for Ireland is 20%, of which two sectors dominate non-Emission Trading System (ETS) emissions: agriculture (44%) and transportation (26%) (Lynch et al., 2016).

The assessment of the environmental impact of land- and sea-based food products can be complex as food production chains involve multiple (and often inter-linking) sub-sectors which may apply different technologies and have different emission footprints. Final food products may have food components from meat, dairy, and grain value chains, which in turn may have inputs from animal feed, fertilizers, and other national and international value chains. The value chains may differentially affect various environmental impact categories (e.g. global warming, acidification, eutrophication, biotic resource use).

A number of different methodologies have been applied to these questions including process-based life cycle assessment (P-LCA), the methodology of the Intergovernmental Panel on Climate Change (IPCC, 2006), and economic input-output life cycle assessment

(EIO-LCA). In P-LCA, all resources and inputs are used, and emissions associated, from raw material extraction and production to end-of-life disposal and waste management are accounted for (Avadí and Fréon, 2015). Input and output flows are quantified. Consequently, P-LCA is often difficult to carry out in reality due to insufficient information and complex interdependencies in the inputs which have to be modelled.

To comply with the GHG emissions reporting requirements of the United Nations Framework Convention on Climate Change (UNFCCC), the IPCC suggested guidelines for GHG emissions accounting (Crosson et al., 2011). The IPCC approach is frequently used to calculate emissions under policy change scenarios. However, the IPCC emissions accounting framework only estimates total emissions generated within national boundaries and does not account for emissions embodied in international trade and transportation.

EIO-LCA analysis combines environmental information with economic data from all sectors within an economy and international trade flows, while addressing shortcomings of traditional processed-based LCA and economic input-output (EIO) analyses. In the EIO-LCA approach, the whole economy is considered as the boundary of the system, with economy-wide interdependencies being modelled as a set of simultaneous linear equations (Joshi, 1999)

An EIO-LCA was conducted to calculate the carbon footprint across the Irish meat supply chains is described in Table 2. The results suggest:

- Poultry meat has the lowest carbon footprint, whereas ruminant meat products (beef, veal, and sheep meat) have the largest impact in terms of carbon emissions, irrespective of the choice of functional unit.
- The carbon footprint of sea fisheries is found to be lower than the carbon footprint of aquaculture when carbon footprint is calculated on monetary, protein and energy use basis. When carbon emissions are expressed per million euro of output or per tonne of protein, aquaculture and sea fisheries have relatively low carbon footprints, outperformed only by poultry meat.
- Aquaculture has the highest output multiplier implying that aquaculture requires more intermediate inputs from other sectors and therefore its carryover effect is relatively greater than other food sectors in terms of production and employment.
- Most value is generated at the processing stage in all food value chains with greater processing value in poultry meat and dairy value chains, and lower value in aquaculture and sea fisheries.

Table 2. Carbon Footprint of Different Land- and Sea-Based Food Products

| | **Output Multiplier** | **kt$CO_2$e per €m of Output** | **t$CO_2$e per tonne of Protein** | **t$CO_2$e per kcal Energy** |
|---|---|---|---|---|
| Aquaculture products | 2.68 | 0.45 | 10.18 | 1.34 |
| Sea fisheries products | 1.75 | 0.34 | 7.68 | 1.01 |
| Beef and veal | 2.58 | 3.47 | 63.18 | 3.72 |
| Sheep meat | 2.15 | 2.44 | 65.87 | 3.87 |

| | | | | |
|---|---|---|---|---|
| Pig meat | 2.40 | 0.75 | 10.19 | 0.41 |
| Poultry meat | 1.81 | 0.31 | 4.35 | 0.38 |
| Dairy products | 2.21 | 1.03 | 28.27 | 1.21 |

Source: Tsakiridis et al., (2020)

### 4.2 Impact Assessment of Novel Technologies

In order to highlight the economic impact assessment capacity of the framework, we develop an engineering model of the implementation of a novel technology, marine-based wave energy devices. The aim of this work is to assess the internal rate of return of an investment in a wave energy facility, specifically, a Pelamis device.

The economic evaluation of wave energy conversion (WEC) devices has been limited by the uncertainty surrounding the true value of existing cost estimates. The framework allows differences in cost conditions to be accounted for within the specified bounds of uncertainty, whilst the variability of output may also be incorporated.

We have used a microsimulation framework to mitigate the effect that this uncertainty may have in policy evaluation, quantifying the likelihood of achieving a given cost estimate. The microsimulation framework comprises a number of steps as outlined in the flow diagram of Figure 5. The microsimulation framework is described in detail in Farrell et al. (2015). The model contains certain and uncertain parameters relating to policy, learning rates and installation size. The model runs Monte Carlo simulations of outputs, as well as operational and capital costs. Probability density functions of risk adjusted rate of return outcomes are derived for different scenarios.

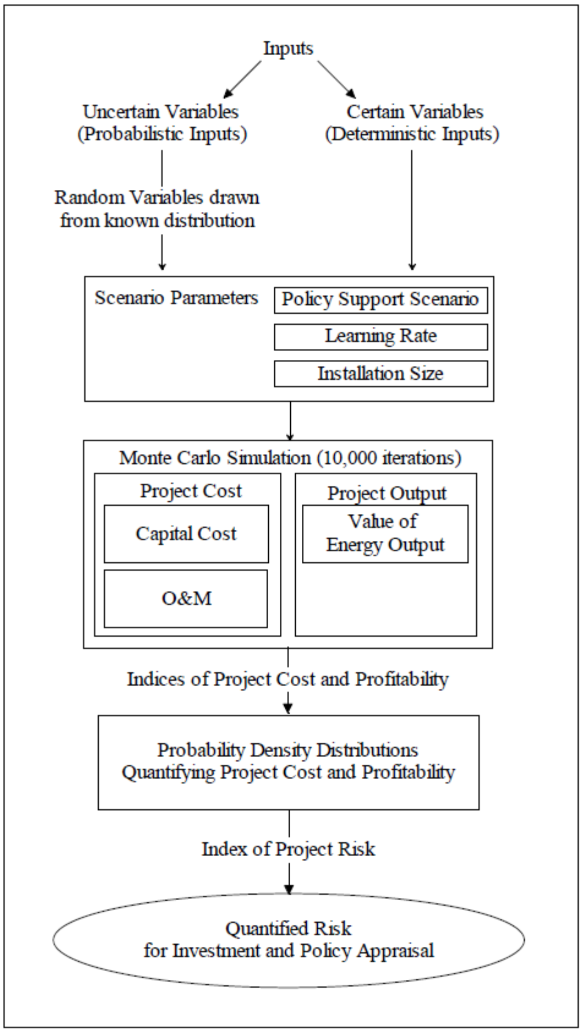


Figure 5. Overview of the Probabilistic Microsimulation Framework

The first goal of this analysis was to quantify cost estimates for a central scenario of deployment. It was found that the expected levelised cost of electricity for 100-unit steel-based installations is €0.203/kWh. The uncertainty surrounding this estimation was quantified using a Conditional Value at Risk (CVaR) approach, which accounts for risk in cost and policy appraisal.

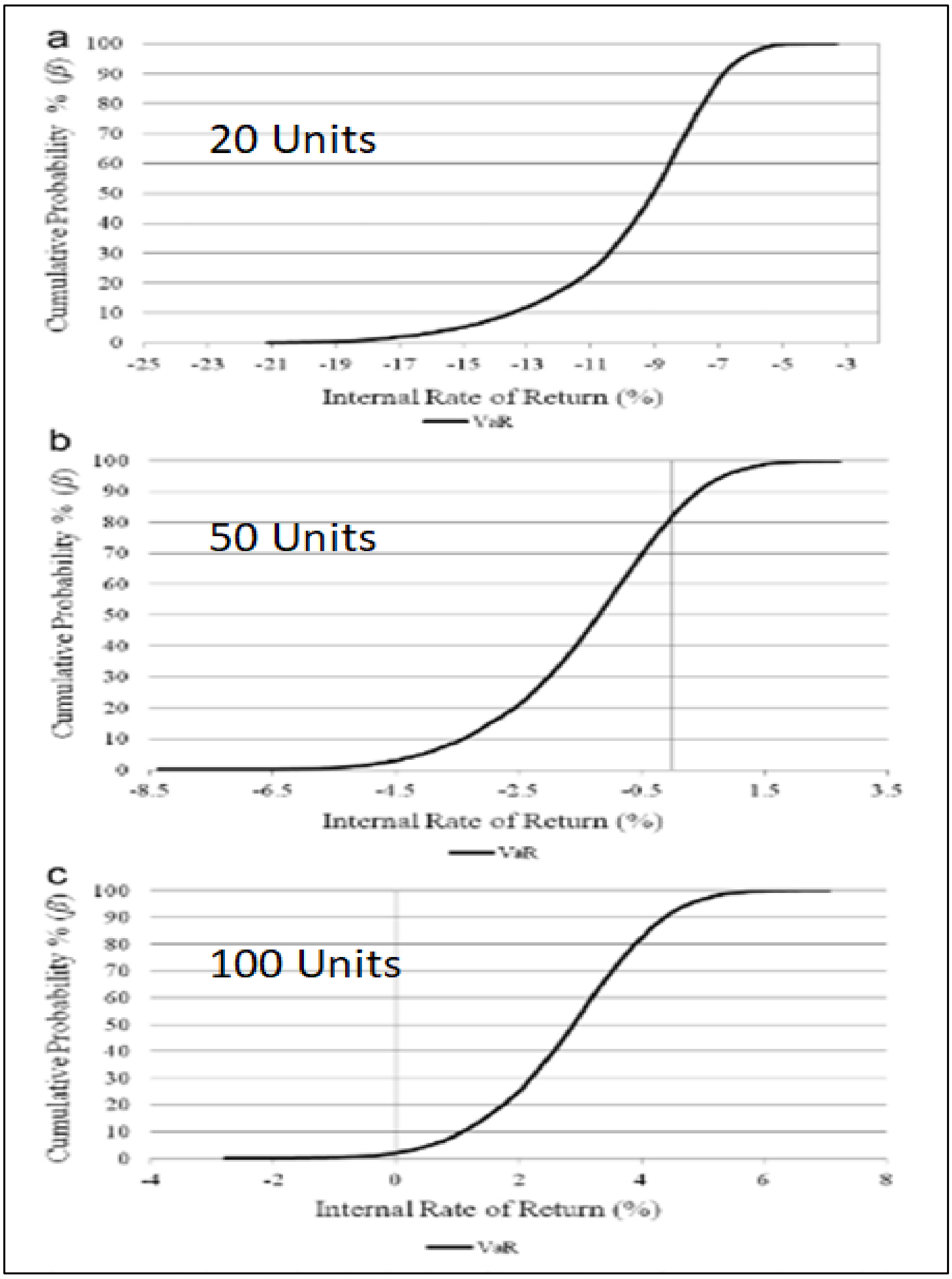


Figure 6. Profitability Distribution of Internal Rate of Return (as a function of scale)

Source: Farrell et al. (2015)

The probabilistic methodology was then applied to quantify the certainty of achieving cost values already quoted in the literature. The results of this analysis provide greater contextual information as to cost estimations quoted to date, allowing policymakers to employ each estimation in the correct context when considering the economic trade-off associated with WEC deployment. Relevant cost estimations were placed in the context of the installation size to which they are likely to refer, with a degree of probability as to their occurrence.

The cumulative probability distribution of the internal rate of return (IRR) is highlighted in Figure 6. The S-shaped curve reports the probability that the internal rate of return, measured with a 6% discount rate is less than the stated rate of return. The analysis is differentiated by the number of devices in the installation. Only 50- and 100-unit installations have the potential to yield a positive IRR when a Renewable Energy Feed-in Tariff (REFIT) of €0.26/kWh prevails. If an IRR of 10% is required for economic viability, then although both 50- and 100-unit installations may offer a positive IRR, there is only a 1% chance that this will exceed the 10% threshold. The analysis shows that, evaluated at the expected (mean) value, a REFIT of €0.49/kWh is required to ensure an IRR of 10% or greater for 20-unit installations, falling to €0.39/kWh and €0.34/kWh for 50- and 100-unit installations.

The methodology and results presented in this analysis are useful for technology developers and investors, as well as for policymakers. For investors, a means to quantify the uncertainty of the investment environment allows for more informed investment decisions. For developers, this model has been applied to determine targets of cost reduction for feasible deployment. Furthermore, using the CVaR methodology allows for potential uncertainties to be incorporated in appropriate targets, such that prudent goals of cost reduction that account for potential cost uncertainties may be defined. Ultimately, the aim is to generalise the framework to be able to apply it to different novel technologies.

### 4.3 Spatial Microsimulation Modelling: Who and Where - Spatial Impact

The spatial microsimulation model SMILE allows one to downscale the analysis to a local district, household, or individual-level scale, providing a spatial dimension to impact assessment. Figure 7 reports the spatial pattern of marine employment by sub-sector. Employment in natural resource sub-sectors such as sea fisheries, aquaculture, and oil and gas exploration are very concentrated in coastal areas. However, marine manufacturing and service sectors have a much greater spatial spread. Seafood processing is generally located nearer to fishing ports, while other marine manufacturing is spatially diffuse. Marine tourism is more concentrated on the coast than other marine leisure services.

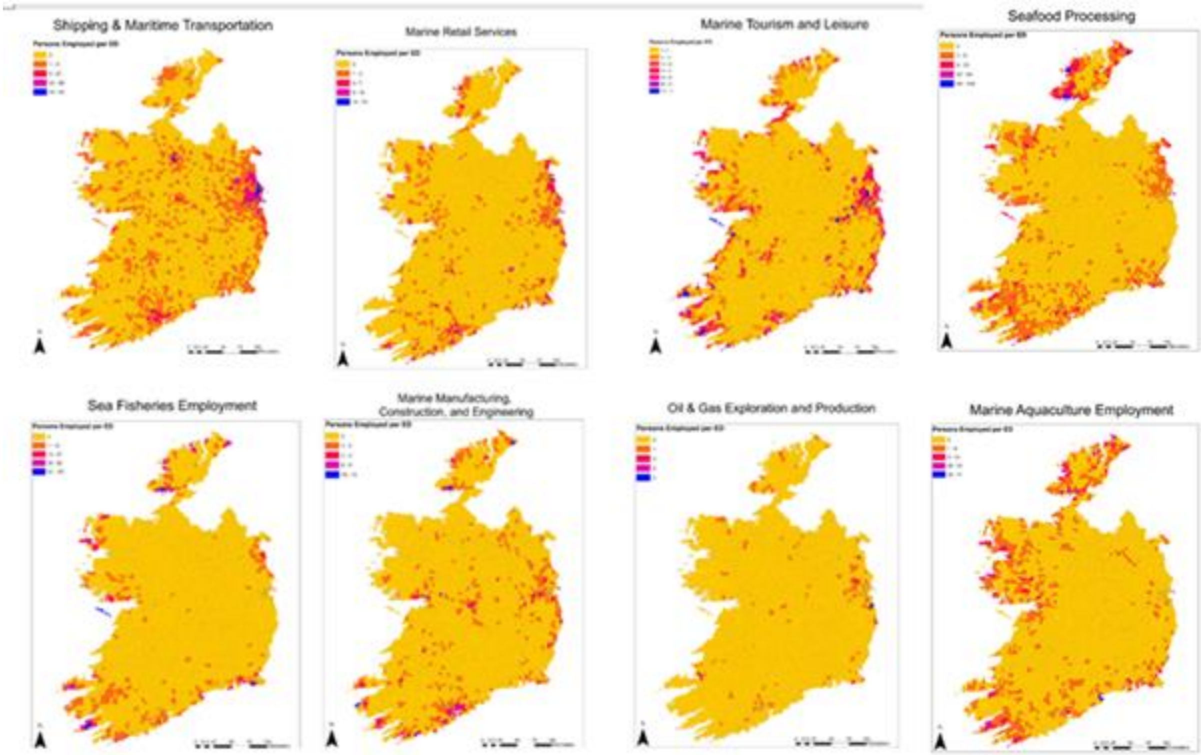


Figure 7. The Spatial Pattern of Marine Employment

Figure 8 describes the average income of EDs with marine sector employees and the spatial distribution of marine income. Although primarily based in coastal areas, some marine services and businesses are located inland. The map of marine sector incomes is quite different to the national distribution, with a much more spatially dispersed income base for the marine sector. The national economy is more concentrated in Ireland's major cities. This emphasises the contribution the sector makes to balanced regional development. These maps describe the spatial and distributional incidence of the marine sector and are a necessary pre-condition for undertaking impact assessment.

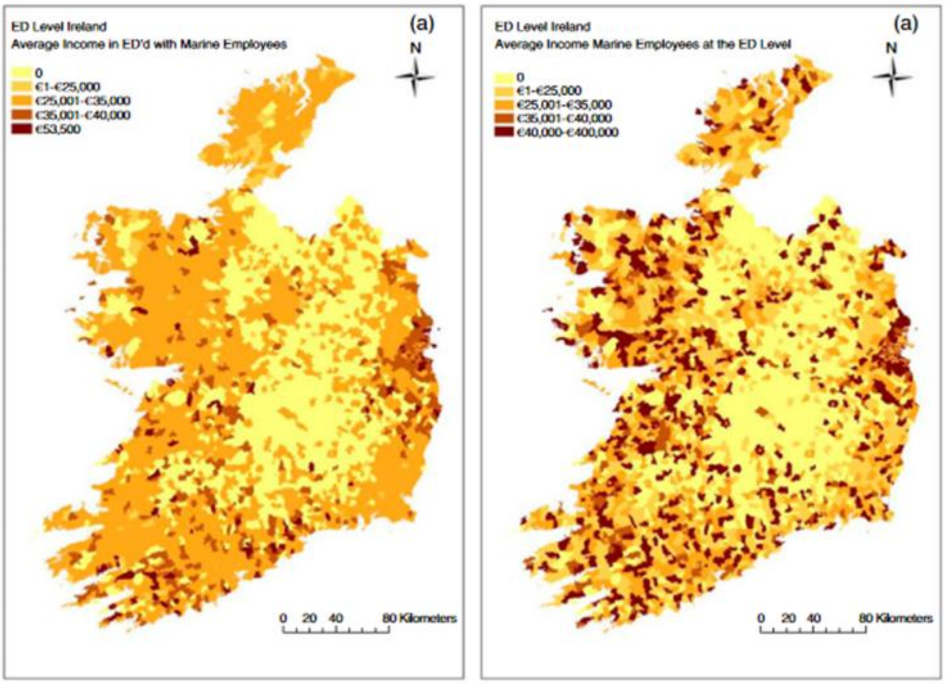


Figure 8. Spatial Economic Footprint

Source: Morrissey et al, (2014)

Using our impact assessment framework, we analyse the spatial impact of a marine renewable energy investment (a 125-unit Pelamis WEC installation) along the west coast of Ireland. An input-output framework is used to model the inputs required (both capital and operational). A supplier database is then employed to model the spatial incidence of these inputs. The labour income of employees in these businesses impacts households elsewhere via a commuting footprint. The Renewable Energy Feed-In Tariff (REFIT) is required to ensure economic viability of these installations. The REFIT subsidy is financed via a public service obligation levy on all consumers, which is a fixed charge on all consumers' electricity bills.

Figure 9 reports the net spatial impact of this investment, together with its financing. The impact is positive in areas where employment is generated. Aside from deployment and manufacturing-related activity, the remaining employment benefit is concentrated in more urban areas and their hinterland, where this added employment has a negligible impact on regional income. It is negative in areas that have to fund the subsidy, particularly for low-income workers. Positive effects accruing from regional employment are undermined by the regressive method through which the scheme is financed, with only concentrated levels of activity providing a means by which a net regional benefit is realised.

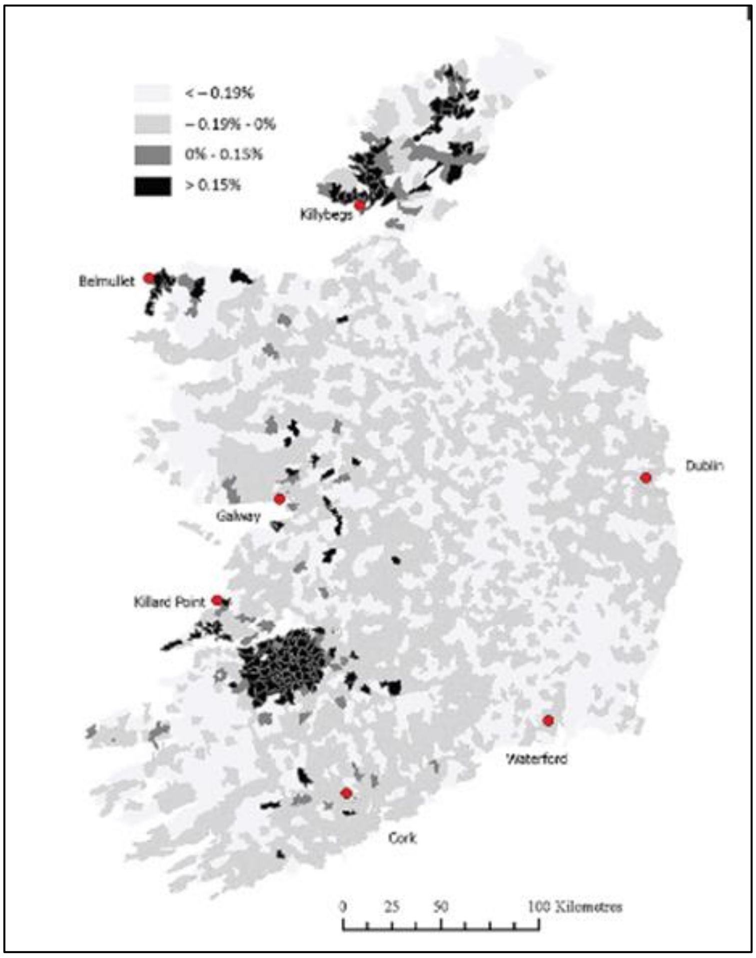


Figure 9. Spatial Impact of Marine Renewable Energy Investment along the West Coast

## 5. DISCUSSION AND CONCLUSIONS

This paper describes the types of analytical strategies used for impact assessment in marine economics, with a specific focus on the utilisation of the methodology in Ireland. The Economic-Social-Spatial-Environmental (ESSE) framework has been developed in order to understand the impact across different dimensions of both government strategies and the development of novel technologies.

The framework has been used to undertake impact assessments of a number of different government strategies in Ireland including:

- Harness Our Ocean Wealth
- Food Harvest 2020
- Food Wise 2025
- Agri-Food 2030

One of the key challenges in undertaking marine economic impact assessment is the availability of appropriate data and both macro and micro level. Marine sectors are often not explicitly reported in either official statistics or in survey data as marine sectors are often subsumed within other sectors. Data collection at present is very time intensive, with manual disaggregation of existing data sources frequently required. Movement to a more streamlined data collection and reporting mechanism, via Marine Satellite Accounts, would allow the release of data more frequently and the field to progress more rapidly.

This framework extends the input-output type models used in marine economics to incorporate micro and spatial dimensions by linking datasets through spatial microsimulation models (van Leeuwen et al., 2017). However, macro-micro links thus far have been relatively crude. In many cases, the impact of macro changes is quite asymmetric at the micro level, requiring more in-depth knowledge of impact pathways.

Lessons for impact assessment from other natural resource economics fields including agriculture and forestry and from the microsimulation field more broadly can point to future developments. First, the micro unit of analysis utilised thus far has been the household. Extending data availability to cover firms would allow for economic analysis to extend below the macro scale to consider issues such as differential productivity and efficiency (Zhai et al., 2012).

Second, while the framework considers the distributional incidence on micro units such as households, it does not consider policy feedback effects. This is relatively limited at both macro scale, with infrequent use of CGE models within the sector, and at the micro level, where structural econometric models with policy feedback loops are rare. Third, sub-national input-output analyses have typically used relatively unsophisticated methods to assign spatial trade. We know from survey data (O'Donoghue, 2021) that spatial patterns vary by sector and by location. Going beyond location quotients by using a micro-based approach in collecting surveys (O'Donoghue et al., 2014) would allow the collection of information on the spatial origin of inputs and the destination of outputs.

One of the main objectives of developing an impact assessment framework is to facilitate decision making, translating data into information. However, knowledge of the impact is merely one dimension required for decision making. Complex decisions such as reducing the carbon footprint, adopting a new technology or forming industrial policy require not only understanding, but also ideas on how to achieve implementation. A system perspective is therefore required. Input-output models incorporate the value chain system but there are typically more agents involved in implementing a strategic change than those directly involved in generating value. This is particularly the case in implementing a strategy or policy that generates public value in addition to private value. An innovation system describes the wider set of agents or stakeholders involved in delivering impact (Brown et al., 2001) and could be a useful approach to adopt for marine policy implementation.

Figure 10 describes an example of a value chain embedded within an innovation system. An innovation system is "a network of organisations, enterprises, and individuals focused on bringing new products, new processes and new forms of organisation into social and economic use, together with the institutions and policies that affect their behaviour and performance" (Rajalahti et al., 2008). A focus on producers independently of other innovation system actors results in imbalanced and unsustainable value chains (Gereffi et al., 2005; Levidow et al., 2012).

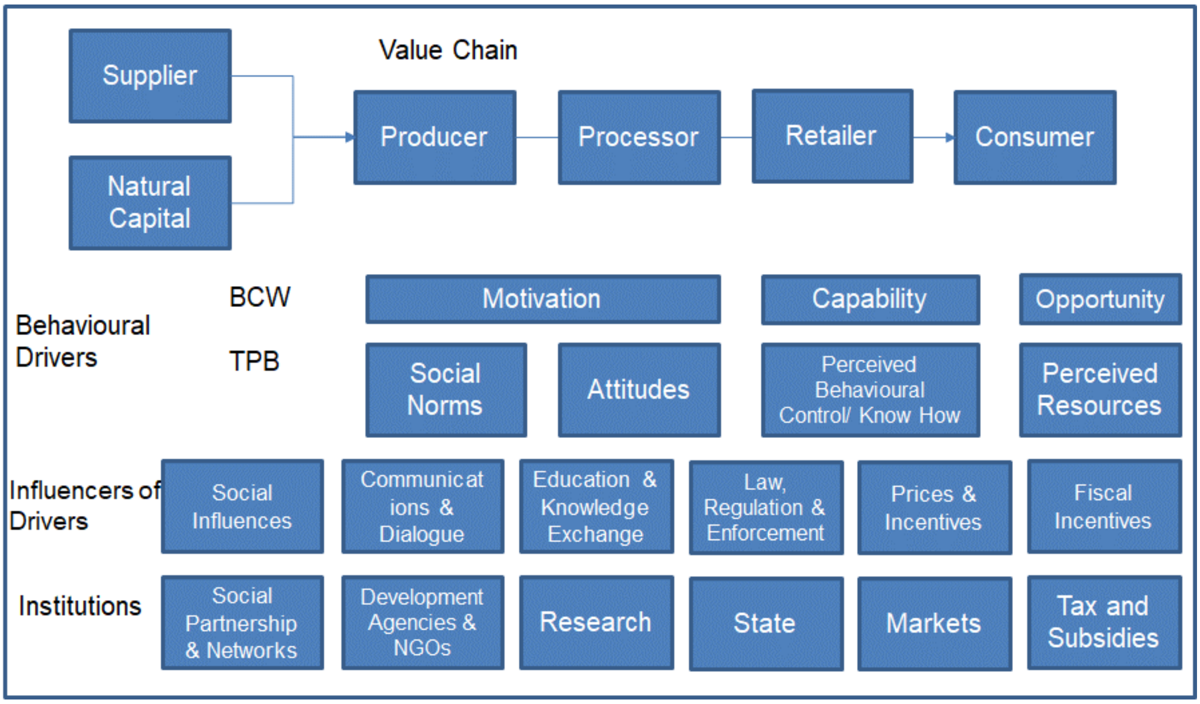


Figure 10. Innovation System

Facilitating behavioural change across a value chain to achieve particular goals also requires changes to behavioural drivers. The theory of planned behaviour (TPB) is a framework that can help explain the drivers of behavioural change (Ajzen, 1991). The TPB framework focuses on:

Perceived behavioural control - know-how, what to do, how to do, and resources (time and money) to be able to do.

Attitudes – why to do or motivation.

Subjective norms – influences from those in whom one trusts.

An understanding of the key variables which are likely to influence value chain actors' perceived behavioural control, attitudes, and subjective norms can be useful in enabling change across the chain and making the results of impact assessments actionable. Incorporating insights from the innovation systems approach and TPB could help to produce impact assessments that can deal with the full complexity of marine policy decisions by providing policy makers with an enhanced understanding of the set of relevant actors and institutions, providing an influence network map of the governance situation involved as well as qualitative and quantitative data about the perceived power and influence of the key marine stakeholders.

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