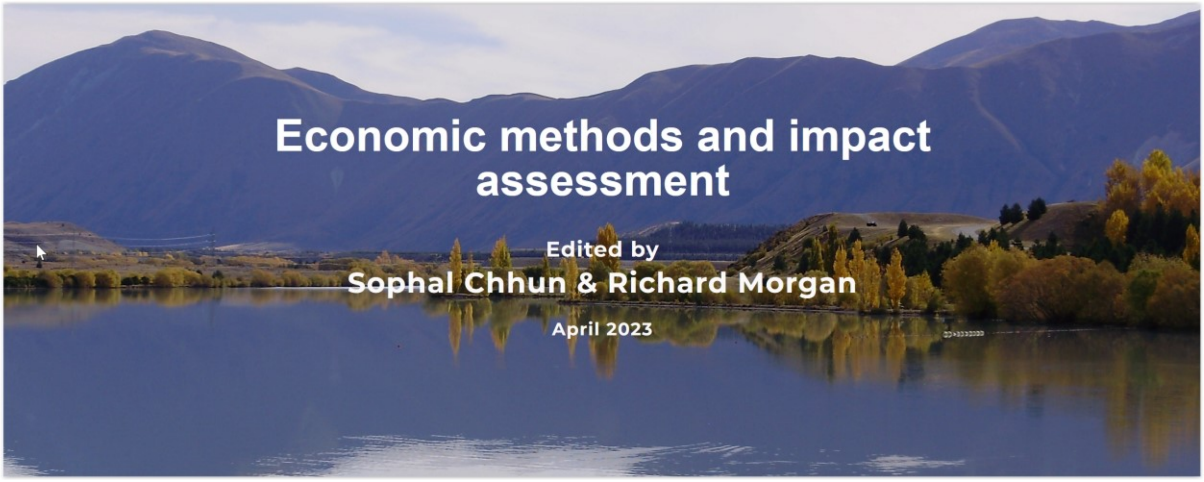


Economic methods in impact assessment: an introduction

Sophal Chhun & Richard Morgan

The nature of economic analysis for resource management

Emma Moran

The state-of-the-art and prospects: economic valuation of ecosystem services in environmental impact assessment

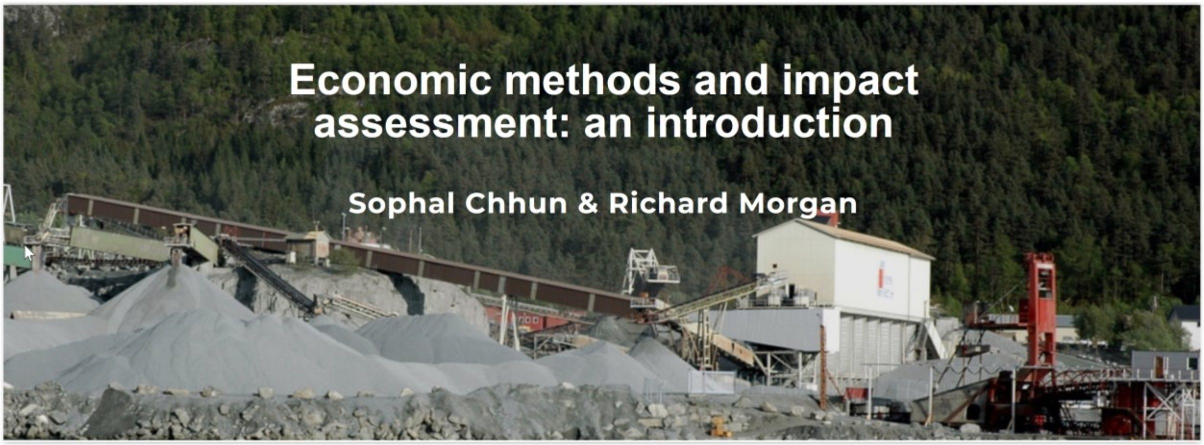
Murray Patterson

Economic impact assessment and regional development: reflections on Queensland mining impacts

Galina Williams

Fonterra's policy on economic incentives for promoting sustainable farming practices

Michael Hide



Introduction

This issue of *Impact Connector* focuses on the use of economic methods and instruments to inform decision making and policy settings within the impact assessment context, to support sustainable development. Among the various economic methods so far developed, economic valuation of the environment, or environmental valuation, has been increasingly used to inform and justify decision-making about resource development. This is a response to traditional cost benefit analyses lacking information about certain values, especially those related to intangible ecosystem services. This leads to such services being perversely treated as having "no value" or at least being given less weight relative to tangible values, such as marketed goods when trade-offs between these values are considered (TEEB, 2010). Economic impact assessment (EcIA) is another method that has emerged, to be used as part of, or in parallel to, conventional impact assessment (IA) in a decision-making process. At a project level, EcIA quantifies, among other things, the socio-economic impacts of proposals such as value added or the contribution to GDP and employment. At a policy level, EcIA can be used in parallel with other IAs to provide quantified costs/benefits of the policy - for example, as part of a 'section 32' report under the RMA in New Zealand.

Despite broad application, there is still lively debate about the relative merits and perceived methodological weaknesses of various economic methods, and how methods and practices can be improved.

Based on "over a decade's experience in applied freshwater economics for local government's regional sector" **Emma Moran** (EM Consulting) examines the application of economic analyses to improve policy making in resource development. The article puts forward pragmatic ideas for more effective economic analyses that will be consistent with the forthcoming environmental and planning legislation, and will help policy makers more effectively avoid the unintended consequences of their decisions.

The editors asked **Prof. Murray Patterson** (Massey University) to reflect on the economic valuation of ecosystem services, a topic for which he has established an international reputation, and comment on the use of such methods in impact assessment. His article provides an excellent primer on the origin and approaches to the challenge of economic valuation of ecosystem services, before considering some of the key issues facing practitioners wanting to assign monetary value to such services, and includes an updated table of ecosystem service unit values for New Zealand land and nearshore ecosystems. Prof Patterson also includes an overview of his work with iwi/hapu to examine the ecosystem services concept in the context of Te Ao Maori, and recognises that cultural and spiritual values, while important components of ecosystem services, should not be subject to economic valuation, but incorporated into decision-making in other ways. The final section of the article makes recommendations for impact assessment practice.

Economic impact assessment is well established in Queensland, Australia, and complements their environmental impact assessment processes. **Dr Galina Williams** uses the mining sector to analyse current EcIA practice. Specifically, she uses a technique called data envelopment analysis (DEA) to compare regions where mining projects were approved to other regions to assess whether EcIA was leading to greater efficiencies in resource use, and identify how practice could be improved to improve resource use efficiency. Dr Williams suggests the current focus in EcIA on employment and income growth should be expanded to include wider socio-economic indicators, to provide a better picture of potential impacts when designing regional economic policies.

In addition to valuation methods and EcIA, it is important to recognise the potential contribution of economic instruments to incentivise behaviour changes. These can be used to promote positive environmental impacts (e.g. payment for ecosystem services), discourage activities that have adverse environmental impacts (e.g. pollution taxes), internalise adverse impacts, or make the polluters responsible for their impacts (e.g. NZ ETS). These instruments provide an important toolkit that may be of use in developing impact mitigation measures, particularly with regard to policy development. It is not practical to cover such a large topic in this issue of the NZAIA Impact Connector – we can only refer you to some excellent works on the topics, including Smith et al. (2013), Hayes et al (2022), Yeldan (2019), Metcalf (2021), and Diaz-Rainey & Tulloch (2018). However, we did want to provide a practical example of the use of an economic instrument to promote better environmental outcomes, and it comes from the dairy industry.

Michael Hide, General Manager for Sustainable Dairying, describes Fonterra's policy on economic incentives for promoting sustainable farming practices in New Zealand. Interestingly, while the financial incentives underpin the first two levels of the programme, the reward for achieving the third level, sustained better environmental practices, is a non-monetary one: the status of being recognised as a top performing dairy operation.

The editors would like to thank the contributors to this issue and trust that our readers are informed and stimulated by reading this issue of NZAIA's *Impact Connector*. Using economic methods in impact assessment can be challenging, so we hope these examples of the application of economic methods and instruments help to overcome some of the challenges.

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Background

For the past 30 years, a system designed to promote the sustainable management of natural and physical resources has been operating in New Zealand under the *Resource Management Act 1991* (RMA). It has become increasingly apparent that the RMA has not fulfilled its early promise (despite initial fanfare) and the New Zealand Government is currently in the process of replacing it with a return to more directive legislation, more akin to the *Town and Country Planning Act 1977.*

As so often is the case, when confronted with problems New Zealand resorts to frenetic legislative activity, drastically changing its regulatory frameworks in order to produce the desired result. Yet almost always the performance does not produce the outcome wished for and we hardly ever measure the effects of our policy changes to see whether their objects have been achieved.

Sir Geoffrey Palmer KC, Address to Resource Management Law Association, 27 September 2013

There are a myriad of reasons for the ultimate failure of the RMA to successfully promote sustainable resource management, many of which are well-documented in the Randerson Report (Resource Management Review Panel, 2020). Yet amongst the reasons generally discussed, little attention appears to be given to the nature of the economic analysis undertaken to inform decision-making (i.e. for a ‘section 32’ report under the RMA).

There have been many examples over the years of where there have been unintended but foreseeable consequences from government policy. A prime example is the loss of undeveloped land of ecological value such as wetlands, particularly in the lowlands, that resulted from the 1950 Marginal Lands Act (Moran and Keenan, 2019) – and was a driver for the Queen Elizabeth II National Trust. The drainage and clearance of land is still treated as ‘improvements’ in New Zealand’s tax system, which perpetuates the wider issue.

More often than not, the economic thinking used to assess the impacts of a possible policy intervention is the same or similar to that which created the issue in the first place, resulting in something of a vicious circle and few solutions.

In general terms, any economic assessment done to inform resource management decisions in New Zealand must be consistent with the principles of the Treaty of Waitangi, and the purpose and principles of the Resource Management Act 1991 and the Local Government Act 2002. If an assessment fails in this respect then its outputs are likely to jeopardise the achievement of policy objectives. Moran (2020)

This article takes the opportunity, created by this turning point in legislation, to put forward pragmatic ideas for improving economic practice 'at the coal face' in the future so it is at least consistent with (rather than undermining) the legislative purpose. These ideas stem from over a decade's experience in applied freshwater economics for local government's regional sector, and before that in biodiversity economics. The article starts by highlighting the importance of setting out a common understanding before touching on what economic analysis is really needed to support policy so it is ultimately effective.

What is the Economy and Economics?[1]

People generally will tell you they 'know' what economy and economics mean, yet when asked to define these terms they can often struggle to put it into words. There are likely to be as many different interpretations as there are people in a room, with some appearing to erroneously conflate it with chrematistics (wealth creation [2]), or even financial accountancy – occasionally because it suits their ends to do so. This 'fuzziness' is evident in the considerable debate that arises when people are set an exercise to categorise things, such as gold, water or honey bees, by placing them on a continuum from 'economy' to the 'natural environment'.
Creating common ground in terms of definitions is a pre-requisite to having a meaningful discussion about the topic. In essence, the economy and economics are broad terms that cover most human activity, whether it be work or leisure.

An economy is the system of activities relating to supply of, and demand for, goods and services in a specific area (including their trade) that helps to 'allocate' resources that are finite or limited (i.e. scarce). No two economies are identical – each one being shaped by a unique set of factors including its culture, laws, history, and biophysical features (landscape, climate and soils).
Economics is how we explain the choices we make (either explicitly or implicitly) through our activities to allocate scarce resources based on their 'utility' (or usefulness) to us, and the implications of these choices for individuals, communities and society.

- Goods and services are all of the flows from our stock of resources (or 'capitals': labour, financial, built, and natural).
- Allocation is the sharing of resources between alternative goods and services – and usually occurs without full information.
- Scarcity is when demand for a good or service is beyond a finite or limited supply.
- Resources are the different forms of capital: natural, human, built, or financial.
- Utility includes 'use' values [3], 'non-use' values, and 'intrinsic' or 'existence' values.

In economics, value is estimated using a range of indicators that are usually (but not always) quantified, with a focus is on those that can be monetised. Neo-classical economics is human-centric, and in practice it is often limited to change in 'use' values where marketplaces exist to trade goods and services. Those values for which there is no market tend to be neglected, meaning most economic analysis is, at best, incomplete – although this is rarely acknowledged.

In a Māori economy, choices are founded on an environmental ethic (Rout, Awatere, Mika, Reid, and Roskruge, 2021). This approach is more akin to the original Greek sense of 'economy' where it denoted the 'rational' management of resources and what was 'rational' was based on ethics (Lesham, 2016).

Skidelsky (2021) devotes an entire chapter to the relatively recent removal of ethics from economics and its consequences.

What is Efficiency?

Similar to 'economy' and 'economics', efficiency is an everyday but much mis-understood term. Efficiency is also erroneously used as a synonym for cost-effectiveness (the latter is simply a measure of the 'cost per unit of output'). Discussions of efficiency tend to use a fairly simple interpretation, focusing on 'productive efficiency' (supplying a good or service in ways that are both technically efficient [**4**] and account for costs). Furthermore, assessments of productivity (and so productive efficiency) are usually incomplete and do not capture externalities. The following expands on this point using fresh water as an example:

Although awareness of water quality issues has improved over recent years, the economy's use of fresh water (both for water takes and to receive by-products as waste) continues to increase in Southland and elsewhere in New Zealand. One reason is that standard assessments of productivity do not usually include an economic activity's use of natural resources over the longer term. In other words, they are partial assessments of productivity, and do not necessarily reflect sustainability. Where an activity's use of water is not accounted for, and it impacts on other values, then all of the community is, in effect, subsidising that activity. This is the case regardless of the economic sector being considered (e.g. agriculture, forestry, manufacturing, tourism or local government).

Moran, Pearson, Couldrey, and Eyre (2017)

Productive efficiency, which includes technical efficiency, is just one dimension of economic efficiency. Economic efficiency is a complex concept and it includes: 1) allocative efficiency (how well resources are shared between goods and services) and 2) dynamic efficiency (how resources are used in ways that improve wellbeing over time). Allocative efficiency and dynamic efficiency are both extremely relevant to managing resources sustainably and yet are rarely mentioned in any analysis [**5**]. In particular, the use of positive discount rates, which weigh the interests of the present generation over future generations, deserves more attention (Parks and Gowdy, 2013).

What Economic Analysis is Really Needed?

The discussion so far likely makes economic analysis appear to be an overwhelming task. Yet there are pragmatic ways of making it manageable and targeted through accurate scoping, problem characterisation, careful choice of multiple measures, using a mix of qualitative and quantitative tools, and seeking expert advice. It starts with developing a firm economic understanding of a topic, which is a 'necessary condition' for a successful policy response to address environmental issues (Moran, 2022).

Scoping starts with a policy context, which shapes the analysis and narrows the economic question(s) at hand. For instance, where a certain course of action is pre-determined, such as via a national policy statement, then economics is about exploring soundness of alternative ways of achieving it rather than re-litigating the direction itself. In other words, testing 'how' something might happen, which means considering 'who', 'what', 'when', 'where' and how much – rather than 'why'. Scoping includes characterising the specific nature of the relevant sectors and activities within the economy, which is influenced by its environmental settings.

The choice and use of economic metrics is critical. For example, at present, analysis tends to focus on partial economic outputs, rather than overall economic outcomes, without acknowledging the gaps. 'Value-added' (i.e. the income earned in the supply of goods and services) gains a lot of attention despite the fact that it may be owned by those far beyond the economy where the resource use (and its environmental effects) is occurring, and can quickly be translated into debt. Employment is usually one of the most relevant measures for local communities (although it needs to capture owner-operators of small businesses).

Despite this information age, not all the data needed for analysis will be readily available or easily understandable. An important but often under-rated source is expert knowledge and opinion. For example, much of what is known about farm debt sits within the agricultural services sector, but it is not accessible for reasons of commercial sensitivity and personal privacy, and is only held by experts (Moran, McDonald, and McKay, 2022). Such qualitative is just as valid as quantitative information, and both are needed in equal measure for completeness.

Models are useful for scenario testing, but there is a tendency to be overly reliant on them. Not everything needs to be modelled, nor does everything need to be included in a model – many things may more easily be addressed in commentary around the results. A conceptual model based on a robust understanding of a system is a useful alternative to a highly developed mathematical 'black box'. Ultimately, the value of models is not so much in producing 'headline' results as learning what is driving those results and so how the system operates, which lead us to think more intuitively about an issue.

For many situations it is sufficient (and even preferable) to develop a range of case studies and real-world examples, especially because industries are often complex and diverse. The process of selecting case studies is an exercise in problem characterisation (e.g. for municipal wastewater in Part C: Section 1 of Moran, McKay, Bennett, West, and Wilson (2018)). Real-world examples are particularly useful for understanding costs avoided – or the benefit side of an equation – where inferential methods can be unsatisfactory. Relevant examples can easily illustrate the costs arising from a deteriorated environment (i.e. damage costs) and the costs of fixing it (i.e. remediation costs [6]) are more tangible (Moran, 2019).

The purpose of economic analysis is not to either be dismissive of, or to overly emphasise, the possible implications – it is to learn and so improve policy solutions by minimising the implications and avoid the unintended consequences. Economics is about providing balance (and, in fact, has similarities to tightrope walking), context and perspective - being what science refers as an 'honest broker' (Pielke, 2007). Gluckman, Bardsley, and Kaiser (2021) offer ten recommendations for effective brokerage that are just as relevant to economics. However, all of this effort is wasted unless it is also communicated effectively to a general audience, which unfortunately tends to be an under-rated skill.

So often, the analysis 'needle' appears to get stuck on what is in front of us now and miss how a situation might reasonably be expected to play out in the future. It also fails to recognise the inconsistencies in the system and so works at cross-purposes. If any new legislation is ultimately going to be successful in promoting sustainable resource management, then our approach to economics has to change with it too. Until this occurs, economic analysis will continue to be our Achilles heel.

Notes

[**1**] This section is based on Moran (2020) and workshops held with Environment Southland's Council and Regional Forum for Freshwater. The Regional Forum was a National Objectives Framework (NOF) process for the National Policy Statement for Freshwater Management 2020: https://waterandland.es.govt.nz/regional-forum

[**2**] From the Greek *khrēmatizein* to make money (*khrēma* meaning money).

[**3**] In general terms, use values can be either consumptive or non-consumptive.

[**4**] Technical efficiency is when resource use is minimised for each unit of output

[**5**] An analogy based on making toast illustrates how all of these dimensions fit together. Imagine you want to make a piece of toast and there are two methods you can use: a metal fork over a camp fire or an electric toaster. Each method has a different mix of resources (e.g. labour, energy, tools) and, depending on the situation, one method or the other will use fewer resources for a given output of toast. This calculation is about the technical efficiency of making toast. Imagine that, as well as the different mixes of resources used in the two methods, you have to also think about costs. The addition of costs shifts the calculus to being about productive efficiency. While imagining all of this toast-making you remember that you also need coffee, and realise that you need to share the use of a resource (maybe it is energy) between the two in a way that makes you as content as possible. Calculating the perfect balance between toast versus coffee is about allocative efficiency. Finally, and having had breakfast, you realise that if you put some resources into developing new tools or growing more raw ingredients you may be able to have more toast or better coffee in the future and be even more content overall. This calculation is about dynamic efficiency across a period of time. A useful reference for further reading is the Australian Productivity Commission (2013).

[**6**] The latter are described here as remediation rather than restoration costs because once an environment has been changed then returning it to a former state can be all but impossible, particularly once ecological thresholds have been crossed

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Introduction

Ecosystem services is one of the fastest growing academic literatures. Based on bibliographic analysis using the Web of Science, Gangahagedara et al. (2021) found that the number of publications and citations have both grown exponentially since 2000. Even before 2000 there were important publications by Daily (1997) who perhaps for the first time provided a wider audience with an understanding of the ecosystem services concept, while Costanza et al. (1997) undertook the first comprehensive valuation of global ecosystem services which attracted worldwide attention from academia and the media. Before these landmark publications, de Groot (1987) was the first to lay down the foundations for the modern analysis of ecosystem services.

Classification and definition of ecosystem services

It would be reasonable to assume that the concept of ecosystem services actually applies to "ecosystems". But that said, as Nahlik et al. (2012) point out, this is not necessarily the case, as it tends to be used as a catch-all phrase to cover any benefits (to humans) from nature or the environment. For example, Boyd and Spencer (2007) define ecosystem services as the "components of nature, directly enjoyed, consumed or used to yield human well-being" and similarly de Groot et al. (2002) define ecosystem services as "the capacity of natural processes and components to provide goods and services that satisfy human needs directly and indirectly".

Many authors, such as Liquette et al. (2013), in developing classification systems only include ecosystem services that directly impact on human well-being, and exclude any candidate ecosystem services that do not directly contribute to human well-being. For example, 'primary production' (photosynthesis) may be excluded because it doesn't directly contribute to human well-being. The Millennium Ecosystem Assessment framework [1] overcomes this problem by defining 'supporting services' such as nutrient cycling or primary production that 'support' other services which directly contribute to human well-being. As can be seen by Figure 1, the Millennium Ecosystem Assessment (2005) framework explicitly shows how 'supporting ecosystem services' contribute to: (1) 'regulating services ' (services that maintain an environment suitable for human habitation), (2) 'provisioning services' (supply of ecosystem goods such as food, raw material and fresh water) (3) 'cultural services' (non-material services such as recreation, tourism, scientific knowledge and aesthetics).

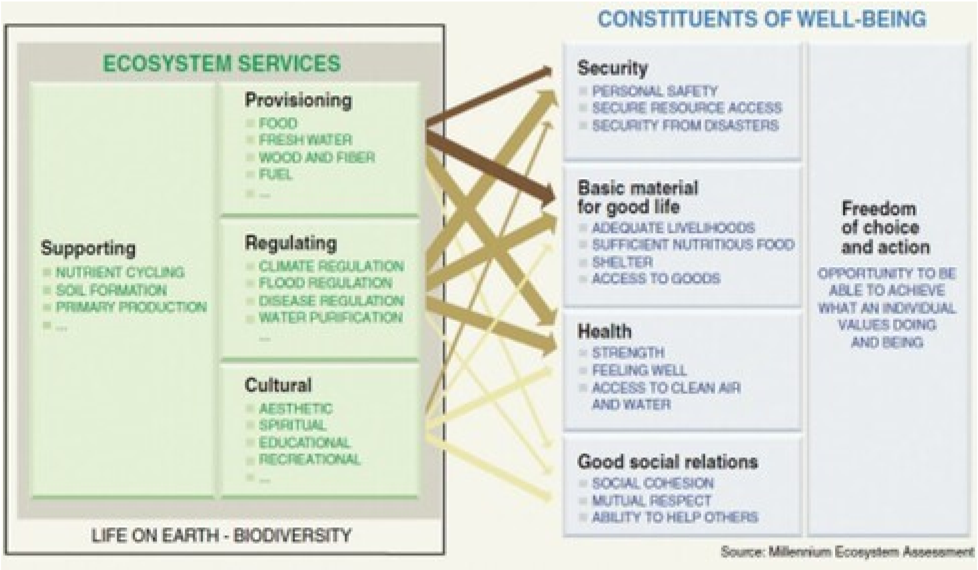

Figure 1. Millennium Ecosystem Assessment Framework

Economic framework for the valuation of ecosystem services

Figure 2 schematically represents a supply and demand curve [**2**] for a substitutable commodity or substitutable ecosystem service. It should however be noted that unlike most market-commodities many ecosystem services are at least to some extent non-substitutable (e.g. climate regulation) and therefore as Costanza et al. (1997) pointed out, the supply curves for these ecosystem services are more like those schematically represented by Figure 3.

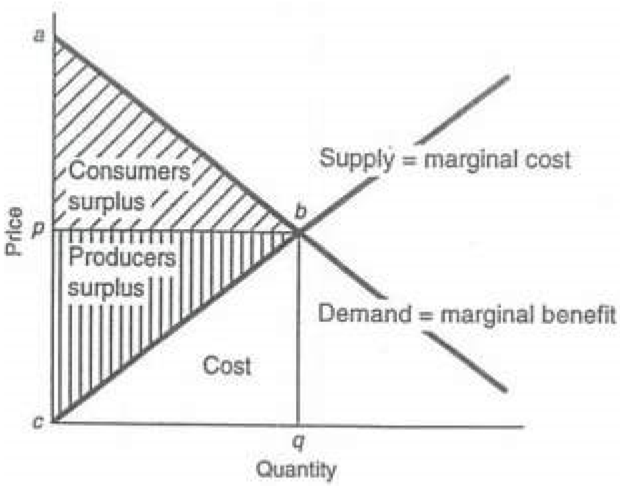

Figure 2. Estimation of Consumers and Producers Surplus for a Substitutable Ecosystem Service

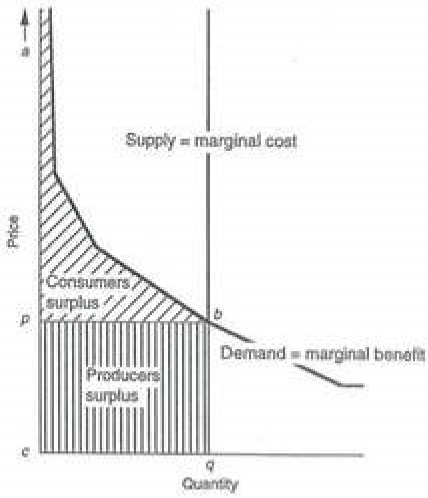

Figure 3. Estimation of Consumers and Producers Surplus for a Non-Substitutable Ecosystem Service

The holy grail is an economic valuation method that captures all of the 'consumer surplus' and 'producer surplus' as outlined by Figures 2 and 3. The 'consumer surplus' is the amount of welfare that a consumer receives over and above the price paid for the ecosystem service; and the 'producer surplus' is the difference between the price that the producers get for a product above the cost of making that product. The sum total of the producers servicing consumer surplus is maximised when the supply and demand curves intersect at the market equilibrium.

Economic valuation methods for quantifying ecosystem services

Many published works on ecosystem services attempt to place an economic value on the ecosystem services using methods outlined below

1. Direct use of market price

The market price method estimates the economic value of ecosystem services that are bought or sold in markets. The market price method can be used to value changes in either quantity or quality of the ecosystem service. Market prices represent the value of an additional unit of a good or service assuming the goods and services are sold through a perfectly competitive market and there is full information available to all participants in the market transaction.

The main advantage of this method is that it is based on 'real life' market transactions, whereas many of the other methods depend for example on hypothetical questions or scenarios being put forward in a questionnaire or an experimental situation. That is, these methods are based on observed data that reflect consumer preferences. From a practical point of view, another advantage is that the method depends on data which is relatively easy to obtain, whether at the micro level data on the market value of individual products (e.g. price of fish), or from macrolevel data that can be uplifted from national economic accounts (e.g. value added by the fishing sector).

There are, however, a number of problematic areas in using market prices. First, market data is only available for a limited number of ecosystem services being particularly applicable to provisioning ecosystem services such as food production. Second, due to imperfections in 'real life' market exchanges, market prices may not reflect the 'true' economic value of ecosystem goods and services. Third, usually the market price method does not deduct the market value of other resources needed to bring the good or service to the market, and thus could be seen to overstate the 'true' market value if this deduction was undertaken.

2. Cost-based methods based on market value

Cost-based approaches are techniques that can be used to estimate the value of an ecosystem service or good by estimating the cost of providing that by artificial means. There are a number of different cost-based techniques including: (1) replacement cost; (2) avoided cost; (3) production function based approaches; and (4) substitute cost.

Cost-based approaches are useful as they provide a rough indicator of economic value, given the availability of data. It is easier to measure the cost of producing benefits rather than measuring the benefits themselves. These methods are also less data and resource intensive, not being dependent on exhaustive surveys. However, expenditures to repair damages or replace ecosystem services are not always good measures of the benefits provided. In some cases, the cost of protective action may actually exceed the value of the of the ecosystem services' benefits.

The **avoided costs method** measures the costs to society 'avoided' due to the presence of an ecosystem service. For example, if ecosystem services provided by wetlands were absent, society would encounter costs associated with 'property loss' (due to the absence of the 'flood control service' provided by wetlands) and 'health costs' (due to the absence of the 'waste treatment services' provided by wetlands'). In this case the presence of wetland ecosystem services is considered to 'avoid' costs associated with property loss and health costs.

The **replacement cost method** values ecosystem services based on the cheapest alternative way of obtaining the same service by using artificial technologies. Woodward and Wui (2001) note that the replacement costs method measures the 'producer surplus'. An example from de Groot (2005) is the ‘natural waste treatment service’ of marshes which can be valued by estimating the cost of replicating the same output as the current ecosystem service with artificial treatment systems.

Production function based approaches model how ecosystem services contribute to the price of a final product in an existing market. The production function approach to valuing ecosystem services is arguably a more ‘dynamic’ approach compared to the more ‘static’ approaches used in the application of other valuation methods – that is, once the production function has been mathematically determined, then various different ‘scenarios’ can be modelled using different input assumptions into the production function. The major advantages of the production function approach are: (1) the market-based data used in the calculations is usually available and robust; (2) the process of building the production function model is relatively straightforward when data is available; (3) production function approaches involve testing of statistical significance of the variables that make up the production function.

3. Revealed preference methods of economic valuation

Revealed preference methods estimate the economic value of ecosystem services that rely on observable market-related behaviour, rather than asking economic agents to make trade-offs among sets of ecosystem services which is the case in ‘stated preference’ approaches That is, revealed preference methods of nonmarket valuation use existing markets that are related to the ecosystem service. Then in these markets, economic agents are assumed to reveal their preferences through their choices.

Hedonic Pricing Method imputes the price of an ecosystem service by statistically modelling how different attributes and/or levels of an ecosystem service (independent variables) affect a known market price (dependent variable). For example if the ecosystem service of ‘flood protection’ increases (+Δ FP), and this increases house prices (+Δ House Prices), then the value of flood protection increases ($Δ FP), can be imputed. The price of labour (wages and salary) in a given region is also often used in hedonic pricing.

The strength of the hedonic pricing method is that it can estimate the economic value of ecosystem services based on ‘actual choices’ of economic agents. In addition, if for example property values are used to estimate the value of ecosystem services, then another strength of the method will be that property markets are relatively efficient in responding to information.

In practice, property values are often used in the application of the hedonic pricing method, and it may be difficult to always relate the value of ecosystem services to property values as readily as other environmental attributes such as the level of noise. From a technical point of view, problems such as the correlation between independent variables (multicollinearity) often hinders the successful application of hedonic pricing.

Travel Cost Method is arguably the first method to be applied to the nonmarket valuation of environmental goods and services. The travel cost method was first developed by Trice and Woof (1958) and Clawson (1959) to estimate the nonmarket value of recreation. The method assumes the value of a recreational site or its services is revealed by how much people are willing to pay to get there. There are several varieties of the travel cost method: (1) simple zonal travel cost method (using mainly secondary data); (2) individual travel cost method (using a more detailed survey of visitors); (3) a random utility approach using survey and other data.

The often cited advantage of the travel cost method is that it measures actual behaviour, rather than hypothetical behaviour. Another advantage is that it is a relatively easy and cost-effective method of nonmarket valuation, although its scope of application is limited to 'recreation' examples.

4. Stated preference methods of economic valuation

Stated preference approaches simulate a market demand for ecosystem services by means of surveys or behavioural economics methods. Stated preference methods can be used to estimate both 'use' and 'non-use' (passive) phases of ecosystem services where there is no existing or surrogate market from which its value can be deduced.

Contingent Valuation Methods ask individuals to state their Willingness to Pay (WTP) for an ecosystem service or their Willingness to Accept compensation (WTA) for the loss of an ecosystem service. Arguably they are the only methods that can be used to assess non-use (passive) value of ecosystem services, but also can be applied to the use value of ecosystems and their services. Although many resource and environmental economists point out that contingent valuation method surveys are straightforward and easy to apply, many others still contest the validity and reliability of such surveys in eliciting values from survey participants. For example, the wide deviation between WTP and WTA estimates of the economic value of environmental attributes/ecosystem services at the very least questions the validity of such surveys (Brander et al., 2010).

The results of contingent valuation methods very much depend on, and are sensitive to, the design of questionnaires, with differences for example being noted for surveys that start with the initially low and initially high opening bids. Inconsistent results, such as the discrepancy between WTP and WTA as Brander et al. (2010) point out can be due to "faulty questionnaire design or interviewing techniques, trigger behaviour by respondents and psychological effects such as 'loss aversion' and the 'endowment effect'."

Although contingent valuation methods are consistent with (neoclassical) economic theory, even if the above problems in their application can be minimised, perhaps the greatest drawback of these methods are that they can be expensive to apply, placing high demands on often limited budgets.

Choice Modelling involves giving individuals the hypothetical setting, and asking them to choose between alternative bundles of attributes associated with an ecosystem service, of which one attribute has a monetary value. Consequently, individuals make trade-offs when choosing between different bundles and different levels of bundles of ecosystem services. As Brander et al. (2010) conclude choice modelling is more complex to design and implement compared with contingent valuation approaches, but to its advantage it's "… more capable of providing value estimates for changes in specific characteristics or attributes of environmental resource (such as an ecosystem service)". As with contingent valuation methods, choice modelling is costly to apply placing pressure on project budgets, as well as requiring complex experimental designs.

Group Valuation as defined by Wilson and Liu (2008) is an approach to the valuation of ecosystem services "based on principles of deliberative democracy and the assumption that public decision-making should not result from the aggregation of separately measured individual preferences but from an open public debate". Group valuation thus is sometimes considered to be an additional "stated preference" method.

What to do when there is a lack of data?

Ideally in impact assessment, the economic valuation of ecosystem services should be undertaken by using any of, or combination of, the above methods (direct market valuations, revealed preference, stated preference). That said, it is not always possible in impact assessment to do this, especially if the comprehensive valuation of all the ecosystem services is the end goal, simply because it is very unlikely that the project will have the financial resources to undertake such a task. For this reason, economists often use the **benefit transfer** method to uplift valuation data from the ‘study site’ (data source site) and apply it to the ‘policy site’ (site which the estimates are applied to). In such an exercise care must be taken to ensure that economic values that are transferred exhibit similarities between the two ‘sites’, and if not, appropriate adjustments are then made to the valuation data.

Kubieszwski et al. (2013) outline four methods of ‘benefit transfer’ for determining the unit value ($/ha) of ecosystem services, which range from ‘basic benefit transfer’, through to using primary data in conjunction with ‘expert opinion’, statistical analysis of primary data, to finally using ‘spatially explicit functional models’.

Importantly, it should also be noted that with the relatively recent development of ‘machine learning’ and the associated advent of ‘big data’, it is likely that analytical power ‘benefit transfer’ methods will significantly improve. Indeed, Wilcox et al. (2018) and Scowen et al. (2021) report that already there has been significant progress the application of machine learning to quantifying and monetising [3] ecosystem services.

Table 1. Unit Values of Land-based and Nearshore Ecosystem Services in New Zealand (NZ_{2020}$ per hectare)

Ecosystem Service	Ecosystem type											
	Cropping & Horticulture	Agriculture	Intermediate Agriculture-Scrub	Native Scrub	Intermediate Angri-Forest	Forests-Scrub	Forest	Wetlands	Estuarine	Mangroves	Lakes	Rivers
Gas Regulation		23	23	0				867				
Climate Regulation	22			288	290	289	290					
Disturbance Regulation								23,827	1,854	6,100		
Water Provisioning	15	10						103			17,919	17,980
Water Storage & Retention								25,010			6,963	6,989
Erosion Control	89	806	95	402	402	402	403					
Soil Formation		3	33	32	30	32	33					
Nutrient Cycling			238	238	238	238	238		12,102			
Waste Treatment		286	286	285	285	285	286	5,461	1,720		2,183	2,191
Pollination	82	82	82									
Biological Control		76	76	12	13	11	13		244			
Refugia								1,433	415	514		
Food Production	16,835	962	202		200						76	81
Raw Materials		59	40		42		1,346					
Genetic Resources												
Recreation		7	3		118	118	118	1,602	1,244		754	759
Cultural		4	7		5	6	7	5,784	98			

Note:
1. This is not a complete coverage of all ecosystem services for each ecosystem type.
2. Derived from Patterson and Cole (2013) and updated.

1. New Zealand data sources

Based primarily on using data from Patterson and Cole (2013) ‘unit values’ ($ per hectare) of nearshore and land-based ecosystem services in New Zealand were updated (refer to Table 1). These ‘unit values’ are spread across 12 different ecosystem types. The main use of these broadscale ‘unit values’ is when they are combined with spatial data, the indicative economic value of the ecosystem services for a farm, catchment, region or even for New Zealand as a whole can easily be calculated. Apart from these broadscale data outlined in Table 1, there are other potential sources of data in New Zealand – for example, Yao and Kaval (2011) provide an excellent summary of nonmarket valuation research in New Zealand covering the period 1974 to 2005.

2. International Data Sources

The most common comprehensive database that can be utilised is the Ecosystem Services Valuation Database (ESVD) compiled by the Foundation for Sustainable Development (2021). The ESVD currently contains over 6,700 value records from over 950 studies distributed across all biomes, ecosystem services and geographic regions. The ESVD organises its value estimates and corresponding data in 106 columns, with information on among others: bibliographic details, study site, biome, ecosystem service, valuation method, valuation result in original units, standardised value (United States dollars/hectare/year in 2020 price levels) and review status. The ESVD supersedes the earlier TEEB valuation database of 1,300 data points from 267 case studies on the monetary values of ecosystem services across all biomes containing data up to and including 2010 (van der Ploeg, de Groot and Wang, 2010).

Other international non-market valuation databases which are reviewed and summarised by Clough and Bealing (2022) may also contain useful data that can be used in ecosystem services valuation, although most of these databases were not designed for this purpose.

Iwi/hapū and impact assessment using the ecosystem services concept

The author has extensive experience working closely with iwi/ hapū from 2005 to 2019, participating and leading large-scale, multidisciplinary and cross-institution research programmes – Ecosystem Services Benefits in Terrestrial Ecosystems for Iwi (2005-2009), Manaaki Taha Moana – Enhancing and Restoring Coastal Ecosystem Services (2010-2015), and Oranga Taiao Oranga Tāngata (2016-2019). Based on his experience, the author's observation is that there is no unitary iwi/hapū interpretation of the relevance of the ecosystem services concept. One viewpoint is that the ecosystem services concept can be utilised to inform decision-making, as well as helping iwi/hapū to advocate for their concerns in environmental management settings. For example, an ecosystem services framework was used by the Tahamata Incorporation in the Horohwenua to evaluate three development options for Tahamata Incorporation's dairy farm (Patterson et al., 2018). Another example is the use by Tauranga Moana iwi of data on the economic value of ecosystem services in the Tauranga harbour as evidence in the Environment Court [4]. At the other end of the spectrum, Cole and Cole (2016) conclude there is no simple conceptual or linguistic analogue for 'ecosystem services' in Te Ao Māori, and that "… the ecosystem services concept is the product of a very different cosmology and perception of reality, its adoption by iwi Māori in preference to whakapapa, kaupapa and/or oral history would probably achieve very little in the long term." Iwi/hapū researchers such as Harmsworth and Awatere (2013) also argue that the 'cultural ecosystem service' is a misnomer, as all ecosystem services are viewed through a cultural lens. Hence the terminology 'non-material ecosystem services' is preferred.

Discussion

Several motivations for carrying out an economic valuation (monetising) of ecosystem services appear in the literature and increasingly, in public discourse. First, it is argued by 'monetising' ecosystem services they become more 'visible' to decision-makers and stakeholders. For example, at the global scale the economic valuation of ecosystem services by Costanza et al. (1997) attracted worldwide attention in the academic literature and in the media. The study found that global ecosystem services had an economic value of $US 33 trillion in 1994 compared with the global GDP of $US 18 trillion. Similar analyses have been carried out in New Zealand at the regional (Patterson and Cole, 1999a; McDonald and Patterson, 2008) and national levels (Patterson and Cole, 1997b; Patterson and Cole, 2013).

Despite this increased awareness of ecosystem services, there are very few examples of the economic assessment of ecosystem services, being applied to impact assessment (of plans, policies, projects). One such example, was the impact assessment of three development options of the Tahamata dairy farm in Horowhenua, which included three climate change adaptation options: (1) no adaptation; (2) some expansion of wetlands; (3) full expansion of wetlands. In this analysis data on the economic value of 18 ecosystem services was modelled over a 30 year period, to enable the iwi landowners (Tahamata Incorporation), to obtain a more 'comprehensive' picture of impacts of these three options (Patterson et al. 2018).

A second motivation for 'monetising' ecosystem services is that it enables the costs and benefits of policies, plans or projects to be integrated into economic decision-making frameworks. For example, Clough and Bealing (2022) review how impacts (i.e. impacts on water, landscape, biodiversity, community cohesion, heritage values) can be monetised, and hence then incorporated into cost benefit analyses of transport projects in New Zealand. At a broader scale, for New Zealand, the loss of soil ecosystem services due to urbanisation has been quantified to be $1,175 million in 2016 ($13,118 per hectare per year of land loss to urbanisation), and these data have been incorporated into macroeconomic accounts (Patterson et al., 2019). This unit value of $13,118 per hectare per year, could be used to indicatively assess the impact of alternative urban expansion policies/plans for New Zealand cities.

Although, as evidenced by the growth in publications by many economists, ecologists, planners and others increasingly support efforts to include ecosystem services in economic decision-making and impact assessment, the approach is not without its critics. Critics first of all point out that the rigorous economic valuation of ecosystem services is difficult and hence is often impractical and expensive. Usually, due to budget constraints, most studies in the literature (that use stated preference or revealed preference approaches) focus on just one ecosystem service, which is problematic because policies, plans and projects impact on many ecosystem services. Therefore, in order to get a more complete coverage of the value of ecosystem services, analysts resort to using benefit transfer methods which in turn are often criticised as being inaccurate.

Critics of the economic valuation of ecosystem services also focus on the philosophical and methodological underpinnings of the approach. Weger and Pascual (2011) for example, amongst many, argue that values associated with the ecosystems are 'plural' and fundamentally incommensurable. Such critics argue that it is not valid to reduce values, which are embedded in complex social systems, to one metric such as 'willingness to pay'. It has consequentially been argued that it is more appropriate to assess impacts on ecosystem services in terms of a participatory process and resist the temptation to quantify or monetise such impacts. The counterargument is that although this may be the case, in reality in our everyday life we make 'choices' and 'trade-offs' that reflect our 'willingness to pay' for goods and services, some of which are don't have a market value. Others argue that the dynamics of ecological systems is non-linear and are often irreversible, which means that ecosystem services cannot be monetised and treated as if they were 'ordinary commodities' that are substitutable. The counterargument to this is that 'monetisation' is not the same as 'commodification' with the former being concerned with economic valuation which does not necessarily imply that the ecosystem service is 'commodified' in the sense that it can be traded on markets. Yet others argue that for most economic valuation methods, the ecosystem service is 'monetised' based on some concept of 'individually determined' values such as an individual's willingness to pay for one ecosystem service, whereas it is argued based on evidence from social psychology that value formation is a 'social' process involving the interaction of individuals. For this reason, some economists such as Wilson and Howarth (2002) have developed 'deliberative' and 'group valuation' methods.

Even though full economic valuation of ecosystem services may prove to be difficult or incomplete or potentially fraught with methodological issues, simply 'identifying' those ecosystem services that will be impacted on (negatively or positively) can in itself be a useful exercise, and can inform which ecosystem services are most significant and therefore may be priority candidates for 'monetisation'.

Finally, it needs to be acknowledged that not all values associated with ecosystems can be validly reduced to an ecosystem services framework, or be monetised. For example, for cultural and spiritual values, although being part of most 'ecosystem service' frameworks, an economic valuation is not appropriate. It is recommended that rather than trying to place a monetary value on cultural and spiritual values the assessment process be carefully be designed so such values can be incorporated in the decision-making at key junctures (Hardy and Patterson, 2012).

Recommendations for Impact Assessment

It was argued above that impact assessments need to be 'comprehensive valuations', including all of the ecosystem services affected by the policy, plan or project. This will require a change in mindset for many economists, who tend to put a lot of effort into measuring economic value of one ecosystem service/environmental good [**5**] very accurately but ignoring the rest. In this regard, measuring the economic value of ecosystem services, it is therefore recommended that the following strategy be adopted [**6**] [**7**]:

(1) as a first priority, the analyst should use **market prices** to measure the value of ecosystem services. For example, there may be a loss of farm production ('food production' ecosystem service). Market prices are preferred as they directly reflect valuation choices made by economic agents in everyday markets.

(2) as a second priority, when project budgets and data availability permit, the analyst should use **revealed preference** methods. Although **revealed preference** methods are based on 'observable market behaviour' which is advantageous, they are based on the application of statistical methods which may have their limitations/ assumptions.

(3) as a third priority the analyst should use **stated preference** methods but be cognisant that they often overestimate the economic value of ecosystem services as they are based on responses to surveys, even when the researcher attempts to make this survey more 'realistic' as in choice modelling.

(4) the last priority, for those ecosystem services that cannot be measured by the above methods, then as a least preferred option is **benefit transfer** method. It should however be noted that 'benefit transfer' methods are being improved, and increasingly more primary studies are available to draw upon. This means that benefit transfer methods are becoming more acceptable as 'proxies' for the economic value of ecosystem services.

It is highly recommended that the analyst should focus on measuring the highest value/s ecosystem services using market prices, revealed preference methods or stated preference methods, rather than using benefit transfer methods. In order to identify the *highest* value ecosystem services and to get a sense of the *relative magnitudes*, it is recommended that a rapid assessment of the economic value of ecosystem services affected by a policy, plan or project be undertaken by using the 'basic' benefit transfer method, perhaps using the data outlined by Table 1 or data from Cole and Patterson (2013).

Notes

[**1**] One of the advantages of the MEA ecosystem services framework is that if properly used, it avoids the problem of double counting 'supporting' ecosystem services which is an often-cited problem encountered in

ecosystem services research. That is, as is shown by Patterson and Cole (2013) demonstrate when aggregating ecosystem services, 'supporting services' should not be double counted.

[**2**] This supply and demand curves diagram represents the standard neoclassical economics model, which underpins the majority of valuations of ecosystem services that are reported in the literature. There are other valuation methods such as biophysical methods as identified by Gómez-Baggethun et al (2016) and Liu et al, (2021).

[**3**] Whilst recognising there are different meanings attached to the term 'monetising', in this paper it is just convenient shorthand for 'placing a monetary value on something'. It can be noted that this shorthand is increasingly being used in the literature – e.g. by Clough and Bealing (2022).

[**4**] In the project *Manaaki Taha Moana* the economic value of ecosystem services in the Tauranga Harbour, mainly derived from mangroves and seagrass, was estimated to be $ 2013 464 million per annum.

[**5**] Yao and Kaval (2011) point out that of the 92 nonmarket valuation studies in New Zealand most only valued one environmental good.

[**6**] It is difficult to generalise when 'Cost-Based' methods (avoided cost, replacement cost) using market prices should be used. Their limitation is that the calculated cost can may exceed actual 'willingness to pay'. Generally, they are preferred ahead of benefit transfer methods, but behind the rest of the methods. However, this is not always the case.

[**7**] Schematically recommended 'merit order' for using economic methods to value ecosystem services: market prices> revealed preference> stated preference> cost based methods using market prices> benefit transfer.

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nzaia
ngā aho tapu

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Economic impact assessment (EcIA) is an essential part of a broader environmental impact assessment (EIA). While EIA is meant to help policy makers to decide whether the proposed project should be granted an approval to move forward, it is typically used for the information and not being determinative in decision making (Wood & Jones, 1997 & Cashmore et al., 2004). Jay et al. (2007) noted a growing dissatisfaction over the EIAs influence on approval decisions. There is a limited interaction between EIA and planning theory, reducing the efficiency of the EIA process (Lawrence, 2000). Furthermore, McDonald & Brown (1995) stated that providing passive advice to decision-makers is inefficient and ineffective, with EIA not leading to solutions. They emphasized the need of aligning EIA to policy and planning.

The problem is that EIA is not designed to stop bad projects, for several reasons including insufficient scope, vested interests, and poor governance, especially where development is equated with economic growth and jobs (Laurence & Salt, 2018). Fonseca & Gibson (2021) noted that projects are rarely rejected in EIA. For example, in Australia, only 18 projects out of 824 projects (2.2% of projects that required approval) have been denied environmental approval since 2000 (Milman & Evershed, 2015).

Economic policy is an important factor in regional development. In Australia, in general, mining is considered as an activity that brings prosperity to the regions. While the concept of the ‘economic base’, with its focus on export activities, is the most popular among the theories of regional development, most recent theories emphasise the importance of diversification, government intervention and investment in infrastructure, and education in order to facilitate economic growth and reduce regional uneven development (Hadjimichalis & Hudson, 2014 &, Alicja, 2009).

EcIA as a part of EIA can help select the projects that increase the long-term sustainable growth in regions (Williams, 2020). However, in practice, it is usually used to justify the project and reports mostly basic information such as employment and income. Regional sustainability and social equality can be measured using a range of indicators such as income distribution, but those indicators are rarely used in impact assessment. As a result, regional areas are often found to be behind South East Queensland metropolitan region in many socio-economic indicators including health, teenage unemployment, low educational attainment, and high domestic violence (Richards 2016, ABS 2021).

Measuring the efficiency of economic policy that is focused on expansion of mining activities (and therefore, mining project approvals) can play an important role in adjusting such policy and achieving improvements in regional performance. The variations among regions in terms of distance from the efficient benchmark can be identified by using data envelopment analysis (DEA). DEA[1] is a non-parametric technique that can be used to analyse the efficiencies of the regions where projects were approved and compare these regions to other regions in order to identify the best practice performance in the use of resources and to highlight where the greatest gains can be made from improvement in efficiency. Typically, regional infrastructure, and the population of a region are used as inputs, and income distribution, employment rate, labour productivity as outputs (Galiniene & Dzemydaite, 2012; Singh-Peterson et al. 2016 & Rabar, 2013). DEA allows social and economic impacts to be modelled in addition to environmental impacts, thus providing a comprehensive assessment of the proposed project at the regional level.

This paper illustrates the use of DEA with various socio-economic indicators to compare mining and non-mining regions efficiencies in Queensland (one of the mining states in Australia). Higher efficiency in mining regions would mean that the policy encouraging mining investments resulted in better utilised resources and higher outputs.

Queensland is the third largest economy in Australia but the average unemployment in Queensland local government areas (LGAs) was higher and mean wages lower than the national average in 2016-17 (ABS, 2019). The income distribution coefficients showed a large disparity in values among LGAs ranging from 0.18 to 0.42. Figure 1 illustrates the income for 2016 and change in coal mining employment in Queensland between 2006 and 2016.

The purpose of EIA in Queensland is to improve the information (often with recommendations) available to government resulting in the overwhelming majority of proposed projects being approved. One of the compelling reasons for approvals of mining projects is that they bring employment (figure 1) and relatively higher incomes to mining workers compared to other industries. That, though, can add to the already existing income inequality in the region. Income inequality is found to be associated with several negative

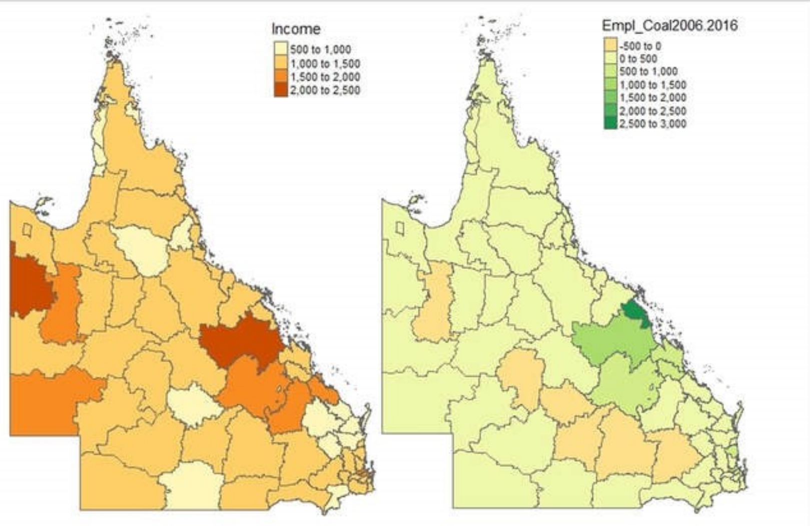

Figure 1. Income (Aus$/week) disparity in Queensland and change in coal mining employment (count of persons aged 15 years and over) in LGAs between 2006 and 2016.

socio-economic outcomes including age specific mortalities, smoking, violent crime, higher expenditure on medical care and police protection (Kaplan et al. 1996). Kawach et al. (1997) suggested that income inequality would result in fewer investments in social capital, while Hill et al. (2012) found a negative association between income inequality and employment resilience.

In Queensland, economic impacts of mining outside of EIA process have been examined at local, regional and state levels (e.g. Williams & Nikijuluw, 2020a, 2020b; De Valck, et al, 2020; Rolfe et al, 2007). The overall results showed that the mining industry created both positive and negative impacts for regional Queensland. Importantly, once externality costs were included, the net present value of coal mining became negative while grazing and conservation options remained positive.

A typical DEA constructs the best practice production frontier [2], which is then used to evaluate relevant efficiency of different units (Farrell, 1957). DEA can have several inputs and outputs in the non-parametric analysis. LGAs are used as units of analysis. An LGA is considered to be inefficient if it generates less output than LGAs with similar resource endowment (Schaffer, et al, 2011). Inefficient LGAs can be thought of the ones that do not utilise resources fully and more focus should be directed to these regions to achieve more efficient outcomes [3].

Inputs and outputs for the study are chosen in line with the regional efficiency literature. This study focuses on income, unemployment, and selected socio-economic indicators such as housing affordability, percent of low-income families and the index of relative socio-economic disadvantage (IRSED) as outputs. In terms of inputs, variables that reflect the resource endowments have been used such as region-specific human capital and infrastructure. Contextual variables are used to account for heterogeneity in regions and include the share of mining and share of agriculture in an LGA's industry structure, population density, industry diversity and population over 65 years old. Summary statistics for the input, output, and contextual variables for the Queensland LGAs are presented in Table 1.

Table 1. Descriptive statistics of variables

	Variable	Mean	Std. Dev.	Min	Max
Inputs	Labour force, FTE	32,265	10,198	47	676,993
	Roads, km	1,950	176	18	7,526
	Secondary education, %	71	1.6	25	100
	Access internet, %	71	1.4	35	88
Outputs	Median family income, $/week	$1,401	$48	$678	$2,777
	Unemployment, %	11	1.2	2.3	50.0
	Low income, %	19.7	1.9	4.4	75.2
	Children in jobless, %	19.6	1.8	0.01	70.6
	Mortgage stress, %	7.3	0.9	0.01	66.7
	Rent stress, %	19.3	1.2	0.01	41.1
	IRSED	886	18	404	1,064
	Over 65 yo, n	8,797	2,389	13	140,681
Contextual variables	Share of mining, %	4.7	0.9	0.0	42.0
	Share of agriculture, %	13.1	1.5	0.0	49.0
	Population density, p/km^2	41.3	14	0.003	842
	Industry diversity, index	0.66	1.09	0.01	6.05

It is interesting to note that the higher percentage of employment in mining does not translate to the higher efficiency for all mining regions (Figure 2). That means that regions can be efficient in utilising their resources without reliance on mining. According to figure 2, some of the most efficient regions have very small percentage of mining employment compared to total employment in LGAs. On the other hand, there is a large proportion of non-mining regions with the efficiency less than average of 0.78. Mining regions tend to have a higher average efficiency (0.84) although not being the most efficient regions.

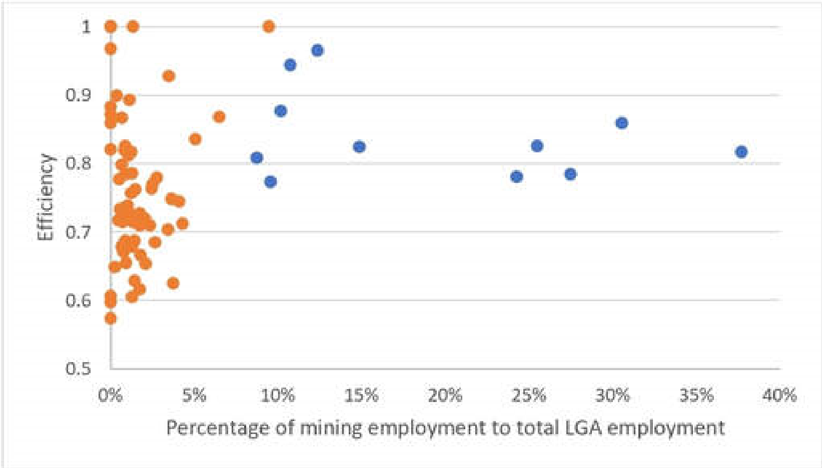

Figure 2. Productive efficiency and mining employment in Queensland LGAs, base model, (blue dots – mining LGAs).

The results of other models are similar to the base model: the efficiency of coal mining regions vary but tend to gravitate towards higher-than-average efficiency. Adding other socio-economic variables such as population density, and industry diversity does not change the mining regions' performance relative to other regions.

More research is needed to identify factors lowering non-mining regions' efficiency and learn from both mining and non-mining regions with higher efficiency.

It is important to note that mining regions were not the regions with the highest efficiency. Therefore, it would be incorrect to infer that all mining projects that bring employment and income to regions are necessarily improving regions' social and economic indicators and utilise resources efficiently. It is important to understand the factors that influence the regional efficiency discrepancy. The analysis using a regression of bias-corrected efficiency scores against a set of contextual variables shows that industry diversity has a highly significant (at 1% level) positive influence on regional efficiency. That is not a surprising result as industry diversity is widely considered a pre-requisite for regional sustainability. Mining share in total employment is not statistically significant, while share of agriculture in the total employment had a positive effect on efficiency and is statistically significant at the 5% level. The proportion of elderly people in the population has a significant negative effect on efficiency. On the other hand, neither population density nor income distribution have a significant effect on efficiency.

The results indicate that policy aimed at growth of a particular sector such as mining does not necessarily improve efficiency of those regions compared to non-mining regions. Further research and more in-depth analysis are needed to understand the reasons behind low performance in some non-mining regions and how to improve efficiency of other regions exposed to the long-term economic policy aimed at growing mining industry.

This example illustrates the importance of a thorough examination of the socio-economic impacts beyond reporting the employment and income from the project during the impact assessment process. Future research can investigate more variables affecting regional development to assist with designing economic policy which is used for conducting EIAs. For example, more socio-demographic variables such as crime rates, hospital admissions, literacy can be used in order to evaluate regional performance. The overall strategic planning should take into account potential negative consequences of reliance on one industry using inputs from various impact assessments. Policy makers can use this approach to refine economic policy to improve regional efficiency.

Notes

[**1**] DEA is used to measure productive efficiency of decision-making units (DMUs). Since it is a non-parametric method, it does not require ex-ante specification of a production or cost-function and, therefore can compare efficiency based on input and output combinations. Most efficient DMUs (eg. those that maximise outputs with given inputs) form the production frontier against which the rest of DMUs is compared.

[**2**] The efficiency is calculated for each DMU as a ratio of the sum of its outputs to the sum of its inputs. Each DMUs efficiency score is calculated relative to an efficiency frontier. Those firms with score less than 100% have the capacity to improve their performance. DMUs located on the frontier are used as benchmarks (Huguenin 2012).

[**3**] This paper uses output-oriented, variable returns to scale DEA model to assess regional performance. The LGAs are assumed to maximise the outputs while holding inputs constant. For example, in the base model, LGAs are assumed to maximise median income and minimise unemployment using available labour force and infrastructure. To adjust for the extreme values the bias-corrected efficiency scores are calculated using contextual variables. This method yields robust and consistent results (Kneip, et al, 2003 & Simar & Wilson, 1998).

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Brief description of the policy

For the 2021/2022 season, Fonterra rolled out a new payment parameter designed to link a portion of the Milk Price to the performance of individual farmers under the voluntary 'The Co-operative Difference' framework. Under this framework, a maximum 10 cents per kilogram of milk solids (kgMS) incentive payment was approved by the Fonterra Board for the 2021/2022 season (this value will subsequently be reviewed annually). The scheme is funded through a re-allocation of the Milk Price. Therefore, the more farmers who achieve under the scheme, the lower the overall Base Milk Price becomes to compensate. The Co-operative Difference framework itself comprises three distinct sequential levels of achievement: Te Pūtake, Te Puku and Te Tihi.

The first level, Te Pūtake, represents Fonterra's strong and sustainable base which is of critical importance when it comes to embodying Fonterra's strategy focus around creating sustainable value. In order to achieve Te Pūtake, farmers must achieve a number of requirements under four focus areas: 'Co-op and Prosperity', 'Environment', 'Animals' and 'People and Community'. All farms that meet the requirements for Te Pūtake receive 7c/kgMS and are eligible to achieve the next level, Te Puku.

Te Puku is where Fonterra recognises milk quality excellence. Our farmer shareholders told us they wanted a milk quality framework that was flexible, motivational and fair. In order to deliver this, an entirely new Milk Quality Framework was developed to replace our existing demerit and grading system. Under this new framework, if a farm achieves 30 days with a milk quality rating of 'Excellence' during the season, they receive an additional 3c/kg MS for all milk supplied with an 'Excellence' rating during the season.

The final achievement level, Te Tihi, represents the pinnacle and is an opportunity for us to recognise the leading farmers in the Co-operative. In order to achieve Te Tihi, a farm must meet the two prior levels of achievement and must maintain a Milk Quality Excellence rating for at least 90% of their milk supplied. Unlike the previous levels, there is no direct financial incentive, rather, Te Tihi is all about recognition. As such, farms that achieve Te Tihi receive recognition in the Annual Report, at annual awards events and they also receive Te Tihi branded merchandise and products.

Catalysts for the development and implementation of the policy and supporting science

Ongoing consumer insight research into our key markets consistently highlights a growing demand for waste minimisation, pasture-based production, animal welfare/quality of life and minimum expectations regarding working conditions for employees on farm. It's becoming more and more evident that the environment in which we operate is perpetually in a state of change; shifts in customer and community expectations, regulations and market requirements can all make change challenging and time consuming.

At times, New Zealand sourced Fonterra products sell at a premium. While it is not always possible to highlight exactly why this is the case, the Co-operative's reputation for pasture based, low carbon dairy is likely a major contributing factor. Considering this, in conjunction with regulatory, social and environmental changes, one can reasonably assume that pro-active management of on farm issues is a good foundation for resilient, sustainable and profitable farming. Conversely, a lack of resilience within the farming businesses that supply the Co-operative represents a critical risk for Fonterra itself. Therefore, in order to support the achievement of our strategic goals, add value to our farmers and position Fonterra as the milk supply company of choice for the New Zealand market, the conclusion was reached that the Co-operative had to move beyond relying solely on minimum standards. As a direct response, The Co-operative Difference was launched in 2019 following extensive feedback from farmers in relation to how they would like to see the Co-operative manage issues around on-farm practices that are likely to impact the Co-operative both now and in the future. After all, our ability to generate value and meet our strategic objectives is largely reliant on our farmers being able to produce high quality, sustainable dairy products that are ahead of increasingly ambitious customer, consumer and stakeholder expectations.

The expected impacts of the policy and actual impacts so far observed

At its core, The Co-operative Difference payment is designed to reinforce the importance of change and encourage early adoption of on-farm practices deemed important to support the achievement of our strategy. We want to recognise and reward our leading farmers who already contribute significant value to the Co-operative by going above and beyond while concurrently incentivising other farms to continuously improve their on-farm performance in a number of key areas. Put simply - more 'carrot', less 'stick'. While the stick can be effective, it often comes at the expense of farmer engagement and is more likely to result in minimum compliance, rather than positive and ongoing change.

A healthy, thriving environment is the foundation of a healthy, thriving farm. Through the environment related achievements in The Co-operative Difference, we are focused on being the most emissions efficient and environmentally sustainable dairy provider by reducing our on-farm footprint and working with the environment. We believe healthy freshwater, soil, ecosystems and a stable climate are essential to the long-term success of dairy farming in New Zealand. In the past 10 years, our farmers have invested heavily in environmental protection, installing new effluent systems and fencing 98% of significant waterways. Under The Co-operative Difference, we are taking this a step further; in order to meet the requirements for the 'Environment' achievement for the 2022-2023 season, farms needed a Farm Environment Plan in place in addition to three out of the five following sub-achievements: having a Purchased Nitrogen Surplus below a target level, participate in a product stewardship scheme for on farm plastics and agri-chemicals, have a winter management plan, have no discharge of effluent to water and/or utilise at least 80% of pasture-based feed across the season.

The welfare of our dairy cows is also of paramount importance. We care for our animals and want them to have healthy disease-free lives. Furthermore, our customers, consumers and fellow New Zealanders expect our animals to be healthy and treated with the care and respect they deserve. To help strengthen the relationship between veterinarians and our farmers to improve animal welfare outcomes, The Co-operative Difference 'Animals' achievement for the 2022-2023 season requires farms to have a current Animal Wellbeing Plan, developed with a registered veterinarian, covering: Body Condition Scoring, mastitis, lameness, mortality, antimicrobial resistance, planning for extreme weather events, polled genetics and management strategies for heat stress. We hope this will lead to a significant improvement in relation to risk identification and mitigation of common animal welfare related issues on farm.

Healthy people is one of the three overarching goals set out in the Fonterra strategy. As an industry, dairy is facing challenges in attracting and retaining talent; this along with agriculture's relatively poor track record in regards to health and safety means there is work to do in this space. Furthermore, many of our major customers have minimum expectations about the way that employees are treated on farm. In order to help improve practices and the dairy industry's reputation as a source of employment, The Co-operative Difference 'People and Community' achievement, requires all participating farms to complete the DairyNZ Workplace 360 assessment and achieve 100% on the first of three sections. At the end of the day, empowering people to create goodness for generations is at the core of our Co-op. Workers should be able to work in safe and healthy work environments so they can thrive at work and get home safe every day.

Milk quality is a key driver for Fonterra in many ways as it can impact the flavour, yields, shelf-life and functionality of end products. Our strong reputation for producing high quality products can largely be attributed to the high-quality milk supplied by Fonterra farmers. In recent years, the Co-operative as a whole has made significant progress towards improving the quality of the bulk milk supply. Today, at a global level, Fonterra's milk quality is firmly in the leading pack and our reputation for quality is strong. However, there still remains significant variation in quality across the supply base; if the average milk quality across the Co-operative was the same as our lowest 10%, it would likely cause significant issues including product quality issues, non-conformance with Overseas Market Access Requirements and increases in the cost of quality failure. Previously, all farms that provided milk that met the quality parameters defined in the Terms of Supply were paid the same regardless of the level of quality provided. By providing a financial incentive related to milk quality, The Co-operative Difference achievement seeks to recognise and reward those farms who provide above average milk quality while concurrently encouraging all farmer to improve their performance and meet expectations. We're confident that this will drive continued improvements to milk quality into the future.

Some reflections on the policy implementation and the way forward

The uptake, awareness and engagement we have seen in this inaugural season of The Co-operative Difference payment has surpassed our expectations with over 6000 farms achieving at least Te Pūtake in the 2021/2022 season. The Farm Source division now sees The Co-operative Difference as central to their engagement with Fonterra's farmer base. This is a very encouraging start in relation to ensuring that farmers have the opportunity to adapt to changes at a pace that works for them and their businesses. As time goes on and the levels of achievement across the farmer base improve, the requirements and payments will also likely change and evolve, thereby continuing the incremental improvements to the standard of farming and ultimately create sustainable value for generations to com