nzaia
ngā aho tapu

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).

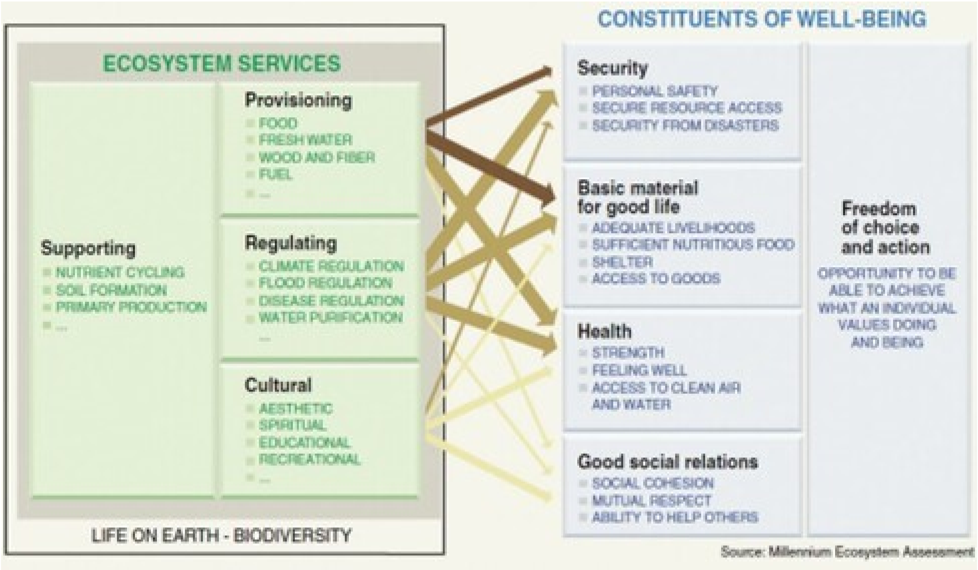

Figure 1. Millennium Ecosystem Assessment Framework

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.

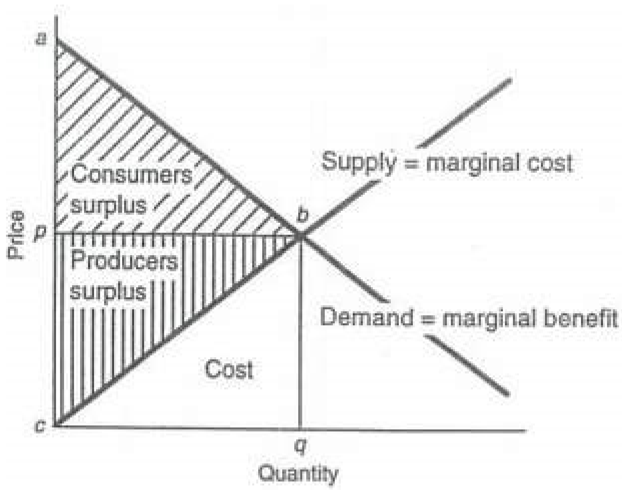

Figure 2. Estimation of Consumers and Producers Surplus for a Substitutable Ecosystem Service

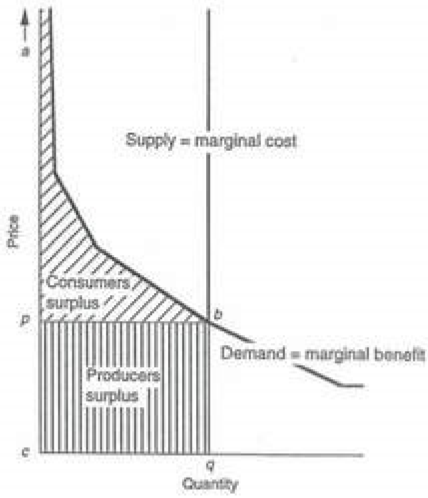

Figure 3. Estimation of Consumers and Producers Surplus for a Non-Substitutable Ecosystem Service

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 costeffective 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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