Motu Working Paper 22-11

Valuing forest ecosystem services in New Zealand

Motu economic & public policy research

Hannah Kotula

November 2022

Document information

Author contact details
Hannah Kotula
Motu Economic and Public Policy Research
PO Box 24390
Wellington 6124
New Zealand
Email: hannah.kotula@motu.org.nz

Acknowledgements
We thank New Forests for commissioning and funding this project. I also thank Arthur Grimes for helpful discussions and feedback.

Disclaimer
The author declares no conflicts of interest.

Motu Economic and Public Policy Research
PO Box 24390 info@motu.org.nz +64 4 9394250
Wellington www.motu.org.nz
New Zealand

 ISSN 1176-2667 (Print), ISSN 1177-9047 (Online).

Abstract

Society depends on services and benefits provided by ecosystems. Yet, many of our actions affect ecosystems in ways that undermine long-term human wellbeing. Although ecosystems provide many services to society, many of these services are not accounted for in land-use decisions. The concept of “ecosystem services” offers a framework for understanding our dependence on nature and can encourage decision makers to consider broader impacts of land-use decisions beyond short-term economic rewards. Furthermore, economic valuation of ecosystem services offers a potential strategy for including the value of ecosystem services in decision making. Here I describe several ecosystem service frameworks and outline how these frameworks can inform land-use decisions, with a particular focus on those involving forests. I then describe methods for valuing ecosystem services. Following this, I provide examples relating to forest ecosystem services and draw conclusions based on existing valuation studies in New Zealand. My intention is to convey how an ecosystem service approach could be used in New Zealand to capture benefits provided by ecosystems that are often not accounted for in land-use decisions.

JEL codes

Q51, Q56, Q57

Keywords

Ecosystem services, nonmarket valuation methods

Summary haiku

Forests provide more
than timber: land-use plans
should reflect this.

Table of Contents

Tables

1 Introduction

Nature supports and sustains life; we depend on the services and benefits provided by nature for our livelihood and wellbeing (Millennium Ecosystem Assessment, 2005).[1] However, many of our actions harm nature, compromising our long-term wellbeing (Foley et al., 2005). The importance of comprehensively accounting for nature in decision-making processes, particularly land-use decisions, is increasingly being recognised (TEEB, 2010). By directly linking human wellbeing with the services and benefits we receive from nature, the concept of "ecosystem services" offers a framework for recognising our dependence on nature, or more concretely, ecosystems.

Ecosystems are constituents of nature; the benefits they bring to society are known as ecosystem services. Ecosystems "combine the abiotic environment with communities of plants, animals, fungi, and microorganisms to form combinations of life forms that control the multitude of natural processes shaping the world around us" (Dasgupta, 2021). Ecosystems are diverse and include watersheds, wetlands, coral reefs, freshwater lakes, rainforests and the oceans. Many services flow from ecosystems: habitat provision, water cycling, timber resources, and inspirational values are all examples of ecosystem services.

Although ecosystems provide us with many ecosystem services, most of these services are not accounted for in conventional land-use decisions which, instead, typically focus on the supply of a single service (e.g., timber provision) (Rodríguez et al., 2006). Important trade-offs in ecosystem service supply can be overlooked when only market-oriented services are considered. A focus on market-oriented services is partly because land-use decisions are often based on economic objectives; many ecosystem services, however, exist outside of the market (Pascual et al., 2010). Determining the monetary value of these ecosystem services offers a potential approach for accounting for these services in land-use decisions (de Groot et al., 2012). In particular, nonmarket valuation methods can be used to estimate the value of a given ecosystem service (e.g., Boyer and Polasky, 2004), noting that the supply of ecosystem services varies across land-use types (Gómez-Creutzberg et al., 2021).

Payments for Ecosystem Services (PES) describe payments made to individuals or communities for maintaining or adopting sustainable land management practices that ensure the provision of ecosystem services (Wunder et al., 2020). The benefits of sustainable land management practices often extend beyond the boundaries of a given property, while the costs of these practices are often local. PES initiatives allow landowners to be compensated for actions that provide regional

[1] Following Dasgupta (2021), 'nature' is treated here as being synonymous with natural capital, the natural environment, the biosphere and the natural world.

and/or global benefits (Wunder et al., 2020). PES initiatives are widely used in some countries and provide important insights into achieving conservation goals.

Forests provide many ecosystem services, and the supply of these services can vary between native and plantation forests (e.g., Dai et al., 2017). Notable services provided by forests include carbon sequestration and the provision of timber. PES initiatives for forest ecosystem services exist both in New Zealand and globally. Further, many studies have estimated the value of a range of forest ecosystem services. Although most of these are international studies, there are a small number of studies that value forest ecosystem services in New Zealand. These studies mostly focus on recreational services and use contingent valuation to estimate the value of these services. These estimates provide a starting point for considering the wider benefits we receive from native and plantation forests.

This paper is organised as follows. In the next section I outline ecosystem service frameworks, and in Section 3 I describe methods that can be used to estimate the economic value of ecosystem services. In Section 4 I describe ecosystem services relating to forests, and in Section 5 I summarise ecosystem service valuation studies relating to forests in New Zealand. The conclusion is in Section 6.

2 Ecosystem services

2.1 Ecosystem service frameworks

There are several approaches to defining and classifying ecosystem services. The Millennium Ecosystem Assessment (MA, 2005) defines ecosystem services as the benefits people obtain from ecosystems. Similarly, The Economics of Ecosystems and Biodiversity (TEEB, 2010) and the Common International Classification of Ecosystem Services, CICES, (Haines-Young and Potschin, 2018) define ecosystem services as the contributions of ecosystems to human wellbeing. Notably, TEEB and CICES distinguish services from the goods and benefits that people subsequently derive from services, whereas MA's definition of ecosystem services encompasses services, goods and benefits. This can lead to double-counting of some services when using the MA framework (further discussed below).

In addition to these ecosystem service approaches, the IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services) presents the related concept of nature's contributions to people (NCP) (Díaz et al., 2015). NCP are all the contributions of living nature to people's quality of life. Living nature is defined as "the nonhuman world, including coproduced features, with particular emphasis on living organisms, their diversity, their interactions among themselves and with their abiotic environment" (Brondizio et al., 2019). NCP extends the concept of ecosystem services by incorporating a more inclusive and interdisciplinary approach (Díaz et al., 2018). MA, TEEB and IPBES are global initiatives, while the CICES was developed by the European Environmental Agency.

Each of the approaches above classifies ecosystem services (or NCP) into broadly similar groups. Table 1 shows the categories of ecosystem services included in each framework. MA, TEEB and CICES include provisioning, regulating and cultural categories (further described below). Additionally, MA and TEEB include a further category (supporting and habitat, respectively). Similarly, IPBES considers three broad categories of NCP: material, nonmaterial and regulating (corresponding broadly to provisioning, cultural and regulating services, respectively). However, a given NCP is not restricted to any one of these categories and can extend across categories, reflecting the overlapping nature of these categories (Díaz et al., 2018). For example, regulation of freshwater and coastal water quality primarily belongs to the regulating category, but extends into the nonmaterial category by providing, for example, recreational value.

Provisioning services are materials obtained from ecosystems, including fibre and freshwater, while regulating services are the benefits obtained from the regulation of ecosystem processes (e.g., regulation of water timing and flows which is defined as "the influence ecosystems have on the timing and magnitude of water runoff, flooding, and aquifer recharge"; MA 2005). Cultural services (e.g., ethical, spiritual, and inspirational values) describe nonmaterial benefits obtained from ecosystems. Unlike the three ecosystem-service approaches (i.e., MA, TEEB, CICES), cultural services are not defined as a separate category in the IPBES. Instead, IPBES recognises that culture mediates the relationship between people and all NCP. In other words, the NCP approach acknowledges that "culture is the lens through which all the elements of nature are perceived and valued" (Brondizio et al., 2019).

Habitat is included as a separate category in the TEEB framework, reflecting the importance of ecosystems in providing habitat for migratory species and in maintaining genetic diversity (TEEB, 2010). Note that habitat provision is included instead as an ecosystem service (or NCP) in the other three frameworks (in the regulatory category for CICES and IPBES, and in the supporting category for MA).

TEEB, CICES and IPBES view processes that are required for the provisioning of all ecosystem services (or NCP) as components of nature (examples of such processes include nutrient cycling and water cycling). In contrast, the MA framework defines such processes as supporting services. By failing to delineate processes from services (and benefits from services, as mentioned earlier), the MA framework risks double-counting some services when valuing a set of services (Fisher et al., 2009; Wallace, 2007). In other words, because supporting services are inputs for all other ecosystem services (e.g., nutrient cycling and soil formation are inputs for crop production), the MA framework can lead to double counting. This risk of double-counting is also present in the other three frameworks: many of the services (or NCP) included in the regulatory category of all frameworks (including the MA) would be more accurately described as processes that lead to services, rather than being services in their own right (Wallace, 2007). For example, erosion regulation is arguably a process rather than a service (Wallace, 2007).

Table 1. Categories included in the four ecosystem service frameworks

	Provisioning	Regulating	Cultural	Supporting	Habitat
MA	✓	✓	✓	✓	-
TEEB	✓	✓	✓	-	✓
CICES	✓	✓	✓	-	-
IPBES	✓	✓	✓	-	-

Note: IPBES refers to the provisioning and cultural categories as material and nonmaterial, respectively, and CICES refers to the regulating category as regulation and maintenance.

MA, Millennium Ecosystem Assessment; TEEB, The Economics of Ecosystems and Biodiversity; CICES, Common International Classification of Ecosystem Services; IPBES, Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

2.2 Biodiversity underpins the supply of ecosystem services

Biodiversity is the variability among living organisms from all sources; this includes diversity within and among species, and within and among ecosystems (Mace et al., 2012). Biodiversity underlies the supply of ecosystem services and influences the productivity of ecosystems (Cardinale et al., 2012). Consequently, biodiversity loss can reduce ecosystem service supply; this view is shared by MA, TEEB, CICES and IPBES. Analogously, Dasgupta (2021) views biodiversity as an enabling asset that gives value to natural capital. Natural capital is defined as “the stock of renewable and non-renewable natural assets (e.g., ecosystems) that yield a flow of benefits to people (i.e., ecosystem services)” (Dasgupta, 2021).

2.3 Ecosystem service frameworks can be used to inform land-use decisions

Ecosystems provide many ecosystem services. However, land-use decisions are often based on maximising the supply of a single service (e.g., crop production) (Robertson and Swinton, 2005; Rodríguez et al., 2006). By accounting for services that are often not considered in land-use decisions, ecosystem service frameworks can encourage decision makers to shift their focus from maximising the supply of a single service to ensuring the supply of a suite of services (Costanza et al., 2014; Guerry et al., 2015).

Land-use decisions are often based on economic objectives. Because many ecosystem services exist outside the market (Barbier et al., 2011), they are often not accounted for in decision making. Having in place monetary incentives that reflect the economic value of ecosystem services offers a potential approach for including the value of these services in decision making (Farber et al., 2002). Economic trade-offs form an important part of policy making; valuation of ecosystems services can help provide economic incentives to sustainably manage ecosystems.

Ecosystem services, by definition, focus on the instrumental value of ecosystems (i.e., their contributions to human wellbeing). Although I acknowledge that ecosystems have intrinsic value, we may choose to focus on the instrumental value of ecosystems (following Dasgupta, 2021) because (i) there are many “systems of thought that go beyond an anthropocentric perspective”, and (ii) if the case can be made to account for ecosystem services in decision making based purely on their value to humanity, then they would be even more worthy of protection if they had intrinsic or sacred status.

Several features of ecosystems contribute to the difficultly of capturing ecosystem services in markets (Dasgupta, 2021). First, ecosystems contain many mobile aspects; oceans circulate, rivers flow, insects fly. Mobility integrates components of an ecosystem and gives ecosystems an element of indivisibility. This means it is often not possible to establish private property rights to many ecosystem services. Second, many ecosystem services, along with the processes underpinning these services, are silent and invisible – this makes it difficult for someone to observe or verify the use others extract from them, and means it can be difficult to trace any damage back to those responsible. Consequently, externalities (both positive and negative) are prevalent in land-use decisions. More fundamentally, many ecosystem processes are nonlinear, and ecosystems can ‘tip’ from one state to another (e.g., a lake can suddenly shift from a clear water state to a turbid water state) (Scheffer et al., 2001). Reversing direction is difficult as ecosystems often display hysteresis; this means restoration is often costlier than conservation.

Although we can view ecosystem services as flows from nature (i.e., natural capital stock), the presence of non-linearities suggests economic valuation should account for both the marginal values from the flows of individual ecosystem services and stock value (i.e., the value of ecosystems which underpin these services). In most situations, marginal valuation rests on an assumption that no irreversible ecosystem changes occur and this assumption only holds at locations distant from tipping points (i.e., only within certain ecological limits).

Importantly, ecosystem service supply can vary across ecosystems. A recent meta-analysis comparing service supply across 25 land covers (as a proxy for land use) in New Zealand found that no single land cover supplies all ecosystem services at a maximal level (Gómez-Creutzberg et al., 2021). This implies supplying multiple services within a landscape will require a mosaic of complementary land covers. Furthermore, Gómez-Creutzberg et al. (2021) found a consistent trade-off in the services supplied by land covers with high-value production versus those with low or no production (note that production refers to economic activity). Land covers with low or no production outperformed land covers with high-value production in supplying several supporting and regulating ecosystem services (e.g., freshwater provision, disease mitigation and regulation of water timing and flows). Note that, except for one[2], all land covers with exotic vegetation had high-value production and all land covers with native vegetation had low or no production. Gómez-Creutzberg et al. (2021) also compared ecosystem service supply between forested and non-forested land covers and found no significant differences between them (with the exception of habitat provision for which forested land covers performed better than those without forest cover); this was true for both native forest and exotic forest.

3 Valuing ecosystem services

The difficultly of capturing ecosystem services in markets (discussed above) means nonmarket valuation methods are often used to determine the value of ecosystem services (McVittie and Hussain, 2013). Nonmarket valuation methods can be used to provide an estimate of the value of a nonmarket good or service and can be divided into two broad groups: revealed preference methods and stated preference methods (Adamowicz et al., 1994). Revealed preference methods (such as hedonic pricing and the travel cost method) rely on observed behaviour or transactions to infer the value of a nonmarket good or service. In contrast, stated preference methods involve surveying people to determine their stated willingness-to-pay for an improvement in the provision of a nonmarket good or service. By estimating the value of a prospective improvement in a nonmarket good or service, stated preference methods can be used to help determine the potential value of implementing, for example, a conservation project. The two main stated preference methods are contingent valuation and choice modelling (Boxall et al., 1996).

Contingent valuation involves directly asking individuals to state their willingness-to-pay to obtain an improvement in a specified good or service. Similarly, choice modelling can be used to determine the willingness-to-pay for marginal changes in the level of ecosystem service provisioning

[2] Low-producing grassland comprises a mix of native and exotic grasslands. Gómez-Creutzberg et al. (2021) assigned this land cover to the ‘no production’ class because it has poor pastoral quality.

(e.g., Hynes et al., 2021). Directly measuring the level of ecosystem service provisioning can be difficult (Eigenbrod et al., 2010; Haines-Young et al., 2012); therefore choice modelling often relies on identifying attributes that capture changes in provisioning (Pearce and Ozdemiroglu, 2002). Survey participants are presented with a number of options, and each option specifies the level of each attribute and the cost of the option.

One example of applying nonmarket valuation methods to ecosystem services is a study by Rivas Palma (2008). To determine the relative value of plantation forest ecosystem services in New Zealand, Rivas Palma (2008) surveyed ten forestry companies, along with a broader group of stakeholders (including consultants, contractors, and Māori groups). By asking survey participants to rank a list of services, the study found that erosion control and water regulation were most highly valued by the respondents. Note that Rivas Palma (2008) separately considered cultural services, along with broader social benefits, and found employment, increased living standard and recreation to be the most valued cultural services and social benefits. Following this, Rivas-Palma (2008) used choice modelling to estimate the willingness-to-pay for marginal changes in the provisioning levels of erosion control and water regulation (note that the study did not determine willingness-to-pay estimates for cultural services and social benefits). Changes in the following attributes were used to capture changes in service provisioning: amount of sediment in water, percentage of land stabilisation, algae in water and level of water flow. The study found that respondents were willing to pay higher amounts for reducing sediment and algae in water compared with improving land stabilisation and reducing water flow level.

Although stated preference methods can be used to estimate the value of ecosystem services (as in the example above), they have several limitations. First, these methods can be influenced by the amount of knowledge respondents have about a given topic (Pascual et al., 2010), noting that the general public may be unfamiliar with ecosystem services. Similarly, these methods can be influenced by the income level of the respondents (Pearce and Ozdemiroglu, 2002). Anchoring effects may also influence estimates if respondents, for example, are presented with examples of low (or high) values before being asked to state their willingness-to-pay. Moreover, it can be difficult to translate individual preferences to some concept of social value, and there is a fundamental difference between saying what you would be willing to pay for something and actually paying for it (Heal, 2000; Pascual et al., 2010).

3.1 Payments for Ecosystem Services (PES)

PES are voluntary transactions in which a user (or a representative of a user, such as government) pays a service-provider for maintaining or adopting sustainable land management practices that ensure the provision of ecosystem services (Wunder et al., 2020). Although ecosystems provide many ecosystem services that extend beyond their boundaries (i.e., spatial externalities are present), the costs of sustainable practices (e.g., natural habitat preservation) often fall on landowners (Green et al., 2018). That is, sustainable practices often come at a high opportunity cost to landowners. PES initiatives can encourage landowners to ensure the provisioning of services, such as climate change mitigation, that provide regional and/or global benefits (Wunder et al., 2020). Note that some PES initiatives pay communal landholders, rather than individual property owners (Alix-Garcia et al., 2018); see Izquierdo-Tort et al. (2022) for an example of benefit sharing in a community-based PES initiative.

3.1.1 Additionality

PES programmes aim to achieve a higher level of ecosystem service provisioning than in the absence of such a programme (referred to as ”additionality”). A number of recommendations have been put forward to achieve this aim.

First, participation should be targeted to high-risk areas that provide high levels of ecosystem service provisioning (thus counteracting adverse self-selection of participants) (Wunder et al., 2020). This is important because both ecosystem service provisioning and environmental risk are often unevenly distributed in space (Wunder et al., 2020). If the opportunity cost of preserving natural habitat is low (e.g., because agriculture on the land is unprofitable), then this land would likely be preserved even in the absence of a PES programme. Thus, making payments to such areas is not optimal from the payer’s perspective of trying to maximise natural habitat cover (Muñoz-Piña et al., 2008). An additional challenge of targeting areas for payments is the potential for spillover effects, specifically leakage (Wunder et al., 2020). Leakage refers to “the extent to which PES participation displaces deforestation and degradation to non-enrolled parcels” (Izquierdo-Tort et al., 2019). A further challenge is that a given area may only have a high level of provisioning for some ecosystem services, introducing trade-offs between various ecosystem services (Locatelli et al., 2014).

Second, payment values for a given ecosystem service should align with both the opportunity costs of providing the service and the value of the service to users (Wunder et al., 2020). A PES programme is unlikely to be effective if the opportunity costs are higher than the payments offered by the programme (Muñoz-Piña et al., 2008). Importantly, if a change in land use is required (rather than a commitment to conserving existing natural habitat), PES initiatives should consider the cost of converting to natural habitat (see the Appendix for a discussion on converting pine forest to native forest in New Zealand). PES initiatives should also consider the long-term opportunity costs

(Wunder et al., 2020). However, compensating providers for opportunity costs may be seen as inequitable when the recipient of the payments is significantly wealthier than the users of the service (Muradian et al., 2013). Finally, effective monitoring and sanctioning of noncompliance is important for a successful PES programme (Wunder et al., 2020).

3.1.2 Capturing spatial variability in ecosystem service provisioning

As mentioned above, ecosystem service provisioning can vary spatially. However, detailed spatial data on ecosystem service provisioning is often lacking and is costly to obtain (Andrew et al., 2015). This means determining site-specific valuations based on service-provisioning levels is often difficult. Therefore, many PES arrangements rely on proxies (such as land-cover condition) that correlate with ecosystem service provisioning (van Noordwijk et al., 2012) or use set payments.

4 Forest ecosystem services

Forests provide many ecosystem services including habitat provision, erosion control, and water cycling. Importantly, native and plantation forests can vary in their supply of services. A recent report (based on Gómez-Creutzberg et al. 2021; described above) compared the supply of 16 ecosystem services between native and exotic forest in New Zealand (Gómez-Creutzberg, unpublished). The 16 ecosystem services compared in this study were: habitat provision, nutrient cycling, soil formation, primary production, water cycling, capture fisheries, freshwater provision, global climate regulation, regional and local climate regulation, regulation of water and timing of flows, erosion control, water purification, waste treatment, disease mitigation, pest regulation, and ethical and spiritual values.

Native forest performed significantly better than exotic forest in freshwater provisioning. However, the study found no significant differences for the other 15 services, potentially due to small sample sizes.

In addition to any differences between native and exotic forests, service supply may differ between secondary forests (e.g., regenerating forest on abandoned agricultural land) and original old-growth forests. Specifically, original old-growth forest tends to supply higher levels of ecosystem services compared to secondary forest (e.g., Mertz et al., 2021), potentially arising from differences in species composition between the two forest types (Chazdon, 2008; Pascarella et al., 2000). Likewise, late-stage secondary forest tends to supply higher levels of ecosystem services than younger secondary forest (Cortés-Calderón et al., 2021; Zeng et al., 2019).

Below I discuss two examples relating to forest ecosystem services in New Zealand, along with examples of forest PES (Payments for Ecosystem Services) initiatives in New Zealand and internationally. I then present a summary table of forest studies in New Zealand that have estimated monetary values for various ecosystem services.

4.1 Forest ecosystem services in New Zealand

4.1.1 Carbon sequestration by native forests

Forests are an important carbon sink. Although carbon sequestration by plantation forests has been well studied in New Zealand, carbon sequestration by native forests has received less attention. Only one carbon look-up table has been provided for indigenous[3] forests by MPI (Ministry for Primary Industries); this table was predominantly derived from measurements of naturally regenerating kānuka/mānuka shrubland[4] and lists the carbon mean annual increment over 50 years as 6.5 tCO_2 ha^{-1} yr^{-1} (Ministry for Primary Industries, 2017; Te Uru Rākau, 2020). However, Kimberley et al. (2021) found that carbon sequestration by planted and managed native forests exceeds this rate. These findings are based on Tāne's Tree Trust Indigenous Plantation Database. Although this is the most comprehensive database on growth rates of planted native stands in New Zealand, many of the stands represented in this database are small and have not been well managed. Carbon sequestration values based on this dataset may therefore underestimate the true value of carbon sequestration in a well-managed native forest. Moreover, native forest has been removed from many lowland areas with fertile soil and if the stands included in Tāne's Tree Trust database mainly occur on poor land, then values may be further underestimated relative to what would occur on more productive land.

4.1.2 Sustainable management of naturally regenerating tōtara

Naturally regenerated stands of tōtara are common on private farms and Māori land in Northland (Scion, 2020). Tōtara tends to regenerate on steep slopes and areas of pasture with poor quality, often where mānuka (or a mix of mānuka and gorse) has established (Bergin and Kimberley, 2014).

This abundance of naturally regenerated tōtara on marginal hill country presents an opportunity to be managed as a future long-term supply of specialty timber. For this reason, the Tōtara Industry Pilot (TIP) project was established in 2018 (and ran until 2020) to determine whether the creation of a tōtara wood products industry in Northland would be viable (Scion, 2020). The

[3] 'Indigenous' is treated here as being synonymous with 'native'
[4] All indigenous forest species are covered by the single generalised forest type: indigenous forest. The indigenous forest carbon table is based on carbon stock values from areas of regenerating indigenous shrublands. Most of these values correspond to mānuka/kānuka as this shrubland type accounts for about 70 percent of the total regenerating indigenous area in New Zealand (Ministry for Primary Industries, 2017).

project was a collaboration between Scion, Te Uru Rakau, Northland Inc., Tāne's Tree Trust and Te Taitokerau Māori Forestry Collective (Steward and Quinlan, 2019). The project found that there is a sufficient amount of naturally regenerating tōtara in Northland to sustain a regional industry, and key next steps have been outlined (Dunningham et al., 2020).

Harvesting tōtara is managed under the Forests Act 1949 through sustainable forest management (SFM) plans and permits, which requires continuous cover to be maintained by removing only single trees or small groups of trees (e.g., 2-5 trees) (Young and Norton, 2017). Members of the Northland Tōtara Working Group have found obtaining either SFM plans or permits for sustainable harvests of farm tōtara on private land to be costly and time consuming. This is partly because (i) the significant variability both within and across tōtara stands makes it difficult to accurately document site-specific forest inventories, and (ii) many totara stands are small; however, the provisions of the Forests Act "cannot be applied to collective management of a tōtara resource across multiple properties." In other words, opportunities for efficiencies (such as reducing the relative permitting cost per unit area to allowable harvest volume) through collective management are not provided for by the Act (Dunningham et al., 2020).

4.2 Examples of PES (Payments for Ecosystem Services) initiatives

4.2.1 Erosion mitigation in New Zealand

One example of a PES initiative in New Zealand was the Erosion Control Funding Programme (ECFP), previously called the East Coast Forestry Project. ECFP was established in 1992 to address erosion in the Gisborne district. The programme had the goal of targeting the worst 60,000 hectares of eroding land in the Gisborne district. However, only about 35,000 hectares were treated through afforestation, reversion or poplar/willow planting (and only about 24,000 hectares of this area was targeted land) (MAF, 2011). The final funding round for ECFP land treatments was held in 2018[5]. A high drop-out rate from grant approval to implementation was a major challenge faced by the programme (MAF, 2011).

While the programme was operating, it offered grants to private landowners to stabilise erodible land, and included the option for native forest regeneration. Grant rates ranged from $1,476 to $2,280 per hectare (depending on distance to port) for afforestation, $1,512 per hectare for reversion, and 70 percent of actual and reasonable costs for poplar/willow planting (MAF, 2011).

[5] https://www.mpi.govt.nz/forestry/funding-tree-planting-research/closed-funding-programmes/erosion-control-funding-programme/

4.2.2 Conserving forest cover to protect hydrological services in Mexico

The PSAH (Payment for Hydrological Environmental Services) programme began in 2003 to address high deforestation rates and severe water scarcity in Mexico. The programme consists of direct payments to landowners with primary (i.e., original) forest cover and is funded through fees charged to water users (Muñoz-Piña et al., 2008). A preliminary evaluation of the programme found that many of the programme's payments have been in areas with low deforestation risk, likely because conserving these areas comes at a low opportunity cost (Muñoz-Piña et al., 2008). A more recent evaluation, focusing on Veracruz state, found similar results: the programme significantly reduced deforestation rates but had limited effectiveness in achieving high additionality (defined above) as most areas enrolled in the programme had low deforestation risk (Von Thaden et al., 2021). This study also considered landscape-scale outcomes (namely, fragmentation and connectivity). The study found that PSAH was not successful in "slowing forest fragmentation or the loss of connectivity in the study regions". The authors caution against focusing on changes in forest cover alone when determining the effectiveness of PES programmes.

5 Ecosystem service valuation studies relating to forests in New Zealand

A comprehensive set of studies valuing a range of forest ecosystem services in New Zealand is not available. Tables 2 and 3 show summary data for the small number of valuation studies from the EVRI (Environmental Valuation Reference Inventory[6]) database and the New Zealand Nonmarket Valuation Recreation database (Kaval and Yao, 2007) that focus on forest ecosystem services in New Zealand.[7][8] These two databases are not a complete set of all published valuation studies. Note that many of the studies in Table 2 estimate the value of an ecosystem service provided by an entire region (which contains many ecosystems including native forest) rather than exclusively for native forest.

Although biodiversity is not viewed as an ecosystem service by the four frameworks (MA, TEEB, CICES, IPBES; as discussed above), I include forest studies that value biodiversity in Table 2 because: (i) biodiversity underpins the supply of ecosystem services, and (ii) biodiversity was the most common focus of New Zealand forest valuation studies in the EVRI database. Kaval et al.

[6] https://www.evri.ca/

[7] The Economics of Ecosystems and Biodiversity (TEEB) valuation database and the Ecosystem Services Valuation Database (ESVD) are two other international valuation databases. Although both these databases contain many valuation studies, neither of these databases contain any studies that focus on forest ecosystem services in New Zealand.

[8] Note that all studies (except one) from the New Zealand Nonmarket Valuation Recreation database are contained in the EVRI database.

(2007b) and Yao & Kaval (2008) both estimate the willingness-to-pay for a programme to plant native trees and shrubs on public and private land, surveying individuals in the Wellington region and in New Zealand, respectively. Lower willingness-to-pay values were estimated by Yao & Kaval (2008), possibly reflecting regional differences in individuals' willingness-to-pay. A further two studies used cost-utility analysis to evaluate the cost efficiency of threatened-species programmes in New Zealand (Cullen et al., 2005, 2001)[9]. Importantly, native biodiversity can be present in plantation forests (see Appendix 1), and several studies have estimated the value plantation forests contribute to native biodiversity conservation (e.g., Yao et al., 2019, 2014)[10].

Following biodiversity, recreation was the second most commonly valued service. Most studies in Table 2 use contingent valuation to estimate willingness-to-pay values. Importantly, willingness-to-pay values may not always align with the value individuals would actually pay (as discussed in 'Valuing ecosystem services' above).

Valuation estimates for a given location may not reflect values at similar locations, for at least two reasons. First, values may depend on the surrounding landscape. For example, the ability of a native forest remnant to provide habitat provision for native species will likely be influenced by the surrounding landscape. Second, the value placed on an ecosystem service will likely depend on the availability of the service within a region. For these reasons, a high degree of caution is recommended when using existing valuation studies to estimate values at new locations that may have materially different characteristics from the study site.

6 Conclusions and recommendations for further research

Nature provides us with many benefits, known as ecosystem services. Our wellbeing and livelihood depend on these services. Yet, many ecosystem services are not accounted for in conventional land-use decisions. Economic valuation offers a potential approach to account for the many benefits we receive from nature in decision-making processes.

Land-use decisions involving forests are particularly important. Valuation studies capturing a diverse array of forest ecosystem services can help decision makers identify, address and balance trade-offs in land-use decisions. In New Zealand, a comprehensive set of studies valuing a diverse

[9] Both of these studies are included in the EVRI database. However, I have not included these studies in Table 2 and Table 3 because they do not provide estimates of the monetary value of biodiversity.
[10] Understorey plant communities in New Zealand pine plantations can vary from diverse to non-existent (McQueen, 1993; Ogden et al., 1997). Temporal changes in understorey light levels (correlated with canopy closure), along with proximity to native forest seed sources, often influence the diversity and composition of understorey communities (Brockerhoff et al., 2003; Forbes et al., 2019). For example, older stands tend to have more diverse understorey communities with a greater number of shade tolerant species and native species (Brockerhoff et al., 2003; Ogden et al., 1997). Furthermore, pine plantations in dry areas typically have few understory plants potentially because native seedlings tend to establish and grow more slowly under dry conditions, and this could allow them to be outcompeted by pine seedlings (Wotton and McAlpine, 2013).

array of forest ecosystem studies is not available. Further valuation studies have the potential to make visible the range of services forests provide us with, beyond market-oriented services such as timber provision, and ensure their inclusion in land-use decisions. However, significant challenges remain in ensuring valuation studies are incorporated into decision-making processes. Specifically, information, resource and capacity gaps can constrain the inclusion of nature's values in decision making. Capacity-building and development, along with collaborations among a range of societal actors, can help bridge these gaps (IPBES, 2022).

References

Adamowicz, W., Louviere, J., Williams, M., 1994. Combining revealed and stated preference methods for valuing environmental amenities. Journal of environmental economics and management 26, 271–292.

Alix-Garcia, J.M., Sims, K.R., Orozco-Olvera, V.H., Costica, L.E., Fernández Medina, J.D., Romo Monroy, S., 2018. Payments for environmental services supported social capital while increasing land management. Proceedings of the National Academy of Sciences 115, 7016–7021.

Andrew, M.E., Wulder, M.A., Nelson, T.A., Coops, N.C., 2015. Spatial data, analysis approaches, and information needs for spatial ecosystem service assessments: a review. GIScience & Remote Sensing 52, 344–373.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecological monographs 81, 169–193.

Beanland, R.A., 1992. Implementation of sustainable resource management: a process for environmental evaluation, Aorangi Awarua case study: a thesis presented in partial fulfilment of the requirements for the degree of Master of Resource and Environmental Planning at Massey University. Massey University.

Bergin, D.O., Kimberley, M.O., 2014. Factors influencing natural regeneration of totara (*Podocarpus totara* D. Don) on grazed hill country grassland in Northland, New Zealand. New Zealand Journal of Forestry Science 44, 1–10.

Boxall, P.C., Adamowicz, W.L., Swait, J., Williams, M., Louviere, J., 1996. A comparison of stated preference methods for environmental valuation. Ecological economics 18, 243–253.

Boyer, T., Polasky, S., 2004. Valuing urban wetlands: a review of non-market valuation studies. Wetlands 24, 744–755.

Brockerhoff, E.G., Ecroyd, C.E., Leckie, A.C., Kimberley, M.O., 2003. Diversity and succession of adventive and indigenous vascular understorey plants in *Pinus radiata* plantation forests in New Zealand. Forest Ecology and Management 185, 307–326.

Brondizio, E., Díaz, S.M., Settele, J., Ngo, H., Guèze, M., Aumeeruddy-Thomas, Y., Bai, X., Geschke, A., Molnár, Z., Niamir, A., 2019. Chapter 1: Assessing a planet in transformation: Rationale and approach of the IPBES Global Assessment on Biodiversity and Ecosystem Services, in: Global Assessment Report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES secretariat, Bonn, Germany.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., 2012. Biodiversity loss and its impact on humanity. Nature 486, 59–67.

Chazdon, R.L., 2008. Beyond deforestation: restoring forests and ecosystem services on degraded lands. Science 320, 1458–1460.

Cortés-Calderón, S., Mora, F., Arreola-Villa, F., Balvanera, P., 2021. Ecosystem services supply and interactions along secondary tropical dry forests succession. Forest Ecology and Management 482, 118858.

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S., Turner, R.K., 2014. Changes in the global value of ecosystem services. Global environmental change 26, 152–158.

Cullen, R., Fairburn, G.A., Hughey, K.F., 2001. Measuring the productivity of threatened-species programs. Ecological Economics 39, 53–66.

Cullen, R., Moran, E., Hughey, K.F., 2005. Measuring the success and cost effectiveness of New Zealand multiple-species projects to the conservation of threatened species. Ecological economics 53, 311–323.

Dai, E., Wang, X., Zhu, J., Xi, W., 2017. Quantifying ecosystem service trade-offs for plantation forest management to benefit provisioning and regulating services. Ecology and evolution 7, 7807–7821.

Dasgupta, P., 2021. The Economics of Biodiversity: The Dasgupta Review. HM Treasury, London.

de Groot, R., Brander, L., Van Der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., 2012. Global estimates of the value of ecosystems and their services in monetary units. Ecosystem services 1, 50–61.

Dhakal, B., Yao, R.T., Turner, J.A., Barnard, T., 2012. Recreational users' willingness to pay and preferences for changes in planted forest features. Forest Policy and Economics 17, 34–44.

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J.R., Arico, S., Báldi, A., 2015. The IPBES Conceptual Framework—connecting nature and people. Current Opinion in Environmental Sustainability 14, 1–16.

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R.T., Molnár, Z., Hill, R., Chan, K.M., Baste, I.A., Brauman, K.A., 2018. Assessing nature's contributions to people. Science 359, 270–272.

Dickie, I.A., Sprague, R., Green, J., Peltzer, D.A., Orwin, K., Sapsford, S., 2022. Applying ecological research to improve long-term outcomes of wilding conifer management. New Zealand Journal of Ecology 46, 1–16.

Dickie, I.A., St John, M.G., Yeates, G.W., Morse, C.W., Bonner, K.I., Orwin, K., Peltzer, D.A., 2014. Belowground legacies of *Pinus contorta* invasion and removal result in multiple mechanisms of invasional meltdown. AoB PLANTS 6.

Dymond, J.R., Ausseil, A.-G., Shepherd, J.D., Janssen, H., 2007. A landscape approach for assessing the biodiversity value of indigenous forest remnants: Case study of the Manawatu/Wanganui region of New Zealand. Ecological Economics 64, 82–91.

Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Heinemeyer, A., Gillings, S., Roy, D.B., Thomas, C.D., Gaston, K.J., 2010. The impact of proxy-based methods on mapping the distribution of ecosystem services. Journal of Applied Ecology 47, 377–385.

Everitt, A.S., 1983. A valuation of recreational benefits. New Zealand Journal of Forestry 28, 176–183.

Farber, S.C., Costanza, R., Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. Ecological economics 41, 375–392.

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecological economics 68, 643–653.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., 2005. Global consequences of land use. Science 309, 570–574.

Forbes, A.S., Norton, D.A., Carswell, F.E., 2019. Opportunities and limitations of exotic *Pinus radiata* as a facilitative nurse for New Zealand indigenous forest restoration. New Zealand Journal of Forestry Science 49.

Gómez-Creutzberg, C., unpublished. Ecosystem service supply from different land covers relative to exotic forests in New Zealand.

Gómez-Creutzberg, C., Lagisz, M., Nakagawa, S., Brockerhoff, E.G., Tylianakis, J.M., 2021. Consistent trade-offs in ecosystem services between land covers with different production intensities. Biological Reviews 96, 1989–2008.

Green, J.M., Fisher, B., Green, R.E., Makero, J., Platts, P.J., Robert, N., Schaafsma, M., Turner, R.K., Balmford, A., 2018. Local costs of conservation exceed those borne by the global majority. Global Ecology and Conservation 14, e00385.

Guerry, A.D., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G.C., Griffin, R., Ruckelshaus, M., Bateman, I.J., Duraiappah, A., Elmqvist, T., 2015. Natural capital and ecosystem services informing decisions: From promise to practice. Proceedings of the National Academy of Sciences 112, 7348–7355.

Haines-Young, R., Potschin, M., 2018. Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure.

Haines-Young, R., Potschin, M., Kienast, F., 2012. Indicators of ecosystem service potential at European scales: mapping marginal changes and trade-offs. Ecological Indicators 21, 39–53.

Heal, G., 2000. Valuing ecosystem services. Ecosystems 24–30.
Hynes, S., Chen, W., Vondolia, K., Armstrong, C., O'Connor, E., 2021. Valuing the ecosystem service benefits from kelp forest restoration: A choice experiment from Norway. Ecological Economics 179, 106833.
IPBES, 2022. Summary for policymakers of the methodological assessment of the diverse values and valuation of nature of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. U. Pascual, P. Balvanera, M. Christie, B. Baptiste, D. González-Jiménez, C.B. Anderson, S. Athayde, R. Chaplin-Kramer, S. Jacobs, E. Kelemen, R. Kumar, E. Lazos, A. Martin, T.H. Mwampamba, B. Nakangu, P. O'Farrell, C.M. Raymond, S.M. Subramanian, M. Termansen, M. Van Noordwijk, A. Vatn (eds.). IPBES secretariat, Bonn, Germany.
Izquierdo-Tort, S., Corbera, E., Martin, A., Lillo, J.C., Dupras, J., 2022. Contradictory distributive principles and land tenure govern benefit-sharing of payments for ecosystem services (PES) in Chiapas, Mexico. Environmental Research Letters 17, 055009.
Izquierdo-Tort, S., Ortiz-Rosas, F., Vázquez-Cisneros, P.A., 2019. 'Partial'participation in Payments for Environmental Services (PES): Land enrolment and forest loss in the Mexican Lacandona Rainforest. Land Use Policy 87, 103950.
Kane, S.J., 1991. Use values of the Hollyford Valley Track: with specific reference to the proposed Haast-Hollyford Tourist Road. University of Otago.
Kaval, P., Yao, R., 2007. New Zealand outdoor recreation benefits. Economics Department, University of Waikato.
Kaval, P., Yao, R., Parminter, T., Scrimgeour, F., 2007a. The value of backyard biodiversity in New Zealand, in: 51st AARES Annual Conference in Queenstown, New Zealand.
Kaval, P., Yao, R., Parminter, T.G., 2007b. The value of native biodiversity enhancement in New Zealand: A case study of the greater Wellington area (Working Paper), Department of Economics Working Paper Series. Waikato Management School.
Kerr, G.N., 1996. Recreation values and kai tahu management: the Greenstone and Caples valleys. New Zealand Economic Papers 30, 19–38.
Kimberley, M., Bergin, D., Silvester, W., 2021. Carbon sequestration by native forest: setting the record straight. Pure Advantage.
Lee, P., Cassells, S.M., Holland, J., 2013. The Non-Market Value of Abel Tasman National Park, New Zealand: A Choice Modelling Application, in: 57th AARES Annual Conference, Sydney, New South Wales.
Locatelli, B., Imbach, P., Wunder, S., 2014. Synergies and trade-offs between ecosystem services in Costa Rica. Environmental Conservation 41, 27–36.
Mace, G.M., Norris, K., Fitter, A.H., 2012. Biodiversity and ecosystem services: a multilayered relationship. Trends in Ecology & Evolution 27, 19–26.
MAF, 2011. Review of MAF Afforestation Schemes.
Marlborough District Council, Department of Conservation, Marlborough Sounds Restoration Trust, 2016. Guidelines for converting pine plantations to native vegetation in the Marlborough Sounds.
McAlpine, K.G., Howell, C.J., Wotton, D.M., 2016. Effects of tree control method, seed addition, and introduced mammal exclusion on seedling establishment in an invasive *Pinus contorta* forest. New Zealand Journal of Ecology 40, 302–309.
McQueen, D.R., 1993. A review of interaction between naturalised woody plants and indigenous vegetation in New Zealand. Tuatara 32, 32–56.
McVittie, A., Hussain, S., 2013. The Economics of Ecosystems and Biodiversity - Valuation Database Manual.
Mertz, O., Bruun, T.B., Jepsen, M.R., Ryan, C.M., Zaehringer, J.G., Hinrup, J.S., Heinimann, A., 2021. Ecosystem service provision by secondary forests in shifting cultivation areas remains poorly understood. Human Ecology 49, 271–283.

Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Synthesis. Washington, DC: Island Press.
Ministry for Primary Industries, 2017. A guide to: Carbon Look-up Tables for Forestry in the Emissions Trading Scheme. Wellington, New Zealand.
Mortimer, R., Sharp, B., Craig, J., 1996. Assessing the conservation value of New Zealand's offshore islands. Conservation Biology 10, 25–29.
Muñoz-Piña, C., Guevara, A., Torres, J.M., Braña, J., 2008. Paying for the hydrological services of Mexico's forests: Analysis, negotiations and results. Ecological economics 65, 725–736.
Muradian, R., Arsel, M., Pellegrini, L., Adaman, F., Aguilar, B., Agarwal, B., Corbera, E., Ezzine de Blas, D., Farley, J., Froger, G., 2013. Payments for ecosystem services and the fatal attraction of win-win solutions. Conservation letters 6, 274–279.
Nghiem, N., Tran, H., 2016. The biodiversity benefits and opportunity costs of plantation forest management: A modelling case study of *Pinus radiata* in New Zealand. Forests 7, 297.
Ogden, J., Braggins, J., Stretton, K., Anderson, S., 1997. Plant species richness under Pinus radiata stands on the central North Island volcanic plateau, New Zealand. New Zealand Journal of Ecology 17–29.
Pascarella, J.B., Aide, T.M., Serrano, M.I., Zimmerman, J.K., 2000. Land-use history and forest regeneration in the Cayey Mountains, Puerto Rico. Ecosystems 3, 217–228.
Pascual, U., Muradian, R., Brander, L., Gómez-Baggethun, E., Martín-López, B., Verma, M., 2010. The Economics of Valuing Ecosystem Services and Biodiversity, in: The Economics of Ecosystems and Biodiversity. Earthscan, London, pp. 183–256.
Patterson, M., Cole, A., 1998. The Economic Value of Ecosystem Services in the Waikato Region, Report prepared for Environment Waikato. Massey University, Palmerston North.
Paul, T.S.H., Ledgard, N.J., 2009. Vegetation succession associated with wilding conifer removal. New Zealand Plant Protection 62, 374–379.
Paul, T.S.H., Ledgard, N.J., 2008. Effect of felled wilding pines on plant growth in high country grasslands. New Zealand Plant Protection 61, 105–110.
Pearce, D., Ozdemiroglu, E., 2002. Economic valuation with stated preference techniques: summary guide. Department for Transport, Local Government and the Regions, London.
Peltzer, D.A., 2018. Ecology and consequences of invasion by non-native (wilding) conifers in New Zealand. Journal of New Zealand Grasslands 39–46.
Rivas Palma, R.M., 2008. Environmental and social values from plantation forests: a study in New Zealand with focus on the Hawke's Bay region. University of Canterbury.
Robertson, G.P., Swinton, S.M., 2005. Reconciling agricultural productivity and environmental integrity: a grand challenge for agriculture. Frontiers in Ecology and the Environment 3, 38–46.
Rodríguez, J.P., Beard Jr, T.D., Bennett, E.M., Cumming, G.S., Cork, S.J., Agard, J., Dobson, A.P., Peterson, G.D., 2006. Trade-offs across space, time, and ecosystem services. Ecology and Society 11, 28.
Sandrey, R.A., Simmons, D.G., 1984. Recreation demand estimation in New Zealand: an example of the Kaimanawa and Kaweka forest parks. Department of Horticulture, Landscape and Parks, Lincoln University.
Scheffer, M., Carpenter, S., Foley, J.A., Folke, C., Walker, B., 2001. Catastrophic shifts in ecosystems. Nature 413, 591–596.
Scion, 2020. Tōtara Industry Pilot Summary.
Scott, N.A., Tate, K.R., Ross, D.J., Parshotam, A., 2006. Processes influencing soil carbon storage following afforestation of pasture with *Pinus radiata* at different stocking densities in New Zealand. Soil Research 44, 85–96.
Statistics New Zealand, 2022. Consumers price index (CPI).
Steward, G., Quinlan, P., 2019. Tōtara industry pilot project: A fresh look at a familiar Northland species. New Zealand Tree Grower.

Te Uru Rākau, 2020. Look-up tables for post-1989 forest land in the Emissions Trading Scheme. Wellington, New Zealand.

Tee, J., Scarpa, R., Marsh, D., Guthrie, G., 2014. Forest Valuation under the New Zealand Emissions Trading Scheme: A Real Options Binomial Tree with Stochastic Carbon and Timber Prices. Land Economics 90, 44–60.

TEEB, 2010. The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB. Progress Press, Malta.

van Noordwijk, M., Leimona, B., Jindal, R., Villamor, G.B., Vardhan, M., Namirembe, S., Catacutan, D., Kerr, J., Minang, P.A., Tomich, T.P., 2012. Payments for environmental services: evolution toward efficient and fair incentives for multifunctional landscapes. Annual Review of Environment and Resources 37, 389–420.

Von Thaden, J., Manson, R.H., Congalton, R.G., López-Barrera, F., Jones, K.W., 2021. Evaluating the environmental effectiveness of payments for hydrological services in Veracruz, México: A landscape approach. Land Use Policy 100, 105055.

Wallace, K.J., 2007. Classification of ecosystem services: problems and solutions. Biological conservation 139, 235–246.

Woodfield, A., Cowie, D., 1977. The Milford Track: valuation estimates of a recreation good. Australian Journal of Agricultural Economics 21, 97–110.

Wotton, D.M., McAlpine, K.G., 2013. Predicting native plant succession through woody weeds in New Zealand. Publishing Team, Department of Conservation.

Wunder, S., Börner, J., Ezzine-de-Blas, D., Feder, S., Pagiola, S., 2020. Payments for environmental services: Past performance and pending potentials. Annual Review of Resource Economics 12, 209–234.

Yao, R., Kaval, P., 2008. Valuing biodiversity enhancement in New Zealand. Department of Economics, University of Waikato.

Yao, R.T., Scarpa, R., Harrison, D.R., Burns, R.J., 2019. Does the economic benefit of biodiversity enhancement exceed the cost of conservation in planted forests? Ecosystem Services 38, 100954.

Yao, R.T., Scarpa, R., Turner, J.A., Barnard, T.D., Rose, J.M., Palma, J.H., Harrison, D.R., 2014. Valuing biodiversity enhancement in New Zealand's planted forests: Socioeconomic and spatial determinants of willingness-to-pay. Ecological Economics 98, 90–101.

Young, L.M., Norton, D.A., 2017. Sustainable tōtara management and biodiversity conservation in Northland. New Zealand Journal of Forestry 62, 23.

Zeng, Y., Gou, M., Ouyang, S., Chen, L., Fang, X., Zhao, L., Li, J., Peng, C., Xiang, W., 2019. The impact of secondary forest restoration on multiple ecosystem services and their trade-offs. Ecological Indicators 104, 248–258.

Appendix: Restoring land to native forest

There is interest in converting land currently planted in pine trees to native vegetation (e.g., Marlborough District Council et al., 2016). However, pine seedlings tend to regenerate and dominate after harvesting if no management intervention is undertaken.

The method used to remove pines can influence subsequent vegetation successions (McAlpine et al., 2016; Paul and Ledgard, 2009). Commercial logging usually destroys most of the native understorey leaving a bare site with a significant pine seed bank (which is generally ideal for pine seedling germination) (Paul and Ledgard, 2009). Similarly, aerial boom spraying often damages understorey vegetation, though some waxy-leaved native species can survive (Marlborough District Council et al., 2016). However, the dead standing trees resulting from aerial boom spraying can provide shade. This can help suppress pine seedling regeneration (pine seedlings are light-demanding). Unlike the previous two methods, ground-based chemical control (i.e., herbicide injection or basal bark application) is targeted: only pine trees die with little damage to native vegetation (Marlborough District Council et al., 2016). This means native vegetation can provide shade and help suppress pine seedling growth. Although this method is expensive, it may be more cost-effective overall as follow-up costs will likely be reduced (Marlborough District Council et al., 2016). The final method involves felling trees and leaving them to rot. This method can facilitate native regeneration (under some circumstances), but it limits site access, can increase fire risk, and can lead to increased levels of weeds (Marlborough District Council et al., 2016; Paul and Ledgard, 2008).

Control methods are rarely successful in completely removing all pine trees, and dominance by other non-native species or reestablishment of pines may follow (Peltzer, 2018). Moreover, long-term ecosystem legacies can remain following pine removal (Dickie et al., 2014). For example, pine trees can change soil chemistry, with the loss of soil carbon being a particularly consistent effect (Scott et al., 2006).

In additional to the removal method used, other factors can influence the success of restoring land to native vegetation. Examples include: site aspect and location (south facing sites tend to have denser native understorey than dry north facing sites), level of weeds within and surrounding the site, and proximity to nearby seed sources (Dickie et al., 2022; Forbes et al., 2019).

Table 2. Valuation studies of forest ecosystem services in New Zealand

These studies are from the EVRI (Environmental Valuation Reference Inventory) database and the New Zealand Nonmarket Valuation Recreation database (Kaval and Yao, 2007). Corresponding notes for each of these studies are provided in Table 3.

Paper	Location	Ecosystem	Ecosystem service	Valuation method	Value	Value unit	Value (2022 $NZ)
(Beanland, 1992)	Aorangi Awarua Forest (North Island)	Native forest	Habitat	contingent valuation	$13.12 per respondent	NZD (1991)	$25.36
(Dymond et al., 2007)	Manawatu/Wanganui Region	Native forest	Biodiversity	landscape approach	$500 million for the region Manawatu/Wanganui	NZD	$697.74 million
(Kaval et al., 2007b)	Wellington Region	Native trees and shrubs	Biodiversity	contingent valuation	$192 (on public land) per resident $208 (on private land) per resident	NZD (2007)	$267.93 (on public land), $290.26 (on private land)
(Nghiem and Tran, 2016)	No specific location	*Pinus radiata* forest	Biodiversity	forest-level optimisation model	$1250 per ha (this is the opportunity cost of conserving biodiversity)	NZD	$1476.54
(Yao and Kaval, 2008)	No specific location	Native trees and shrubs	Biodiversity	contingent valuation	$82 (on public land), $42 (on private land) [per person per year in annual rates]	NZD (2007)	$114.43 (on public land) $58.61 (on private land)
(Yao et al., 2014)	No specific location	Planted forest (mainly *Pinus radiata*)	Biodiversity	discrete choice experiment	$24.18 per respondent	NZD	$28.8
(Yao et al., 2019)	Waikato region	Planted forest (mainly *Pinus radiata*)	Biodiversity	discrete choice experiment	$138,371 for the Waikato region	NZD	$155,667.38
(Dhakal et al., 2012)	Whakarewarewa Forest	Planted forest (redwoods and a mix of other exotic species)	Recreation	travel cost	$34 per visit for walkers, $48 per visit for mountain bikers	NZD	$41.43 for walkers $58.5 for mountain bikers
(Everitt, 1983)	Kauaeranga Valley	Multiple ecosystems*	Recreation	variant of the travel cost method	$100,000 per year for the Kauaeranga Valley	NZD (1981)	$482,720.34
(Kane, 1991)	Hollyford Track	Multiple ecosystems	Recreation	travel cost	$110 per visit	NZD	$212.66
(Kerr, 1996)	Greenstone & Caples Valleys	Multiple ecosystems	Recreation	contingent valuation	$42 per person per trip	NZD (1994)	$78.51
(Lee et al., 2013)	Abel Tasman National Park	Multiple ecosystems	Recreation	choice modelling	$0.64 per additional native bird species per park visitor	NZD	$0.77
(Sandrey and Simmons, 1984)	Kaimanawa & Kaweka Forest Parks	Multiple ecosystems	Recreation	travel cost	$27.16 (see Table 3)	NZD (1984)	$98.83
(Mortimer et al., 1996)	Little Barrier Island	Native forest	Conservation activity	contingent valuation	$37.31 per household per year	NZD	$65.38

(Rivas Palma, 2008)	Hawke's Bay	Plantation forest (mostly *Pinus radiata*)	Erosion control & water regulation	choice modelling	$338.18 per household per year for five years	NZD	$453.69
(Tee et al., 2014)	No specific location	*Pinus radiata* forest	Carbon storage	a real options approach using the binomial tree method	$14,290 per ha	NZD	$17,414.54
(Patterson and Cole, 1998)	Waikato Region	Forest (both native and plantation)			$2,400 per ha per year (for forest ecosystems in the Waikato region in the year 1997).	NZD	$4106.18

Notes:

- *"Multiple ecosystems" includes native forest
- To convert nominal prices to real prices (2022 $NZ), I used the June Consumers Price Index (CPI) from Statistics New Zealand (2022). Note that I assumed values were expressed in dollar values of the year of publication in cases where this information was not provided in the study.
- Kaval et al. (2007b) is included in the EVRI database. However I do not include this paper in this table because this paper is an earlier version of Kaval et al. (2007a), which I do include.
- Woodfield and Cowie (1977) is included in the New Zealand Nonmarket Valuation Recreation database (Kaval and Yao, 2007). Although this paper provides an estimate of the recreational value of the Milford track, I do not include it here in this table as insufficient information is provided in the paper to determine the unit of the estimate.

Table 3. Notes for valuation studies

Paper	Notes
(Beanland, 1992)	$13.12 is the mean amount respondents were willing-to-pay annually to preserve the Aorangi Awarua Forest.
(Dymond et al., 2007)	This is the total economic value.
(Kaval et al., 2007b)	The values $192 & $208 represent the mean amount residents in the Wellington Region would be willing-to-pay annually in their rates for a program to plant native trees and shrubs on public and private land. The paper also provides separate estimates for urban and rural respondents (Table 3 in Kaval et al. 2007).
(Nghiem and Tran, 2016)	Patch clear cutting (of *Pinus radiata* forest in New Zealand) provides biodiversity benefits but comes at an opportunity cost of $1250 per hectare
(Yao and Kaval, 2008)	$82 & $42 are the willingness-to-pay values (per person per year in annual rates) for planting native trees and shrubs. The authors surveyed individuals across New Zealand.
(Yao et al., 2014)	$24.18 is the willingness-to-pay estimate per person for supporting a brown kiwi conservation project. The paper also provides willingness-to-pay estimates for other species in Table 4.
(Yao et al., 2019)	This paper determines the willingness-to-pay to support a brown kiwi conservation programme in planted forests for various regions in New Zealand. The value presented ($138,371) is the annualised conservation cost (NZD per year) for Waikato. The paper provides values for other regions in Table 7.
(Dhakal et al., 2012)	Values are given separately for walkers and mountain bikers.
(Everitt, 1983)	$100,000 is the minimum value for the recreational benefits of the Kauaeranga Valley.
(Kane, 1991)	$110 is the mean willingness-to-spend on additional travel costs to the Hollyford Track.
(Kerr, 1996)	$42 is the weighted mean benefit across all four groups (hunters, anglers, trampers, trekkers) for one trip to the area.
(Lee et al., 2013)	$0.64 is the average value park visitors would be willing-to-pay for the presence of an additional native bird species. The paper also gives WTP estimates for the presence of huts and other features in Table 6.
(Sandrey and Simmons, 1984)	$27.16 is the average consumer surplus per visit
(Mortimer et al., 1996)	$37.31 is the average amount households (in Auckland) would be willing-to-pay to maintain current conservation activities on Little Barrier Island.
(Rivas Palma, 2008)	$338.18 is the willingness-to-pay estimate (per household per year for five years) for improving sediment in water from high to low. The study also provides estimates for other attributes in Table 9.16
(Tee et al., 2014)	$14,290 is the Real Options (flexible harvest) valuation of carbon forestry per ha
(Patterson and Cole, 1998)	$2,400 is the value (per ha per year) of ecosystem services produced by forest ecosystems in the Waikato Region in 1997. This estimate includes both direct and indirect use values.