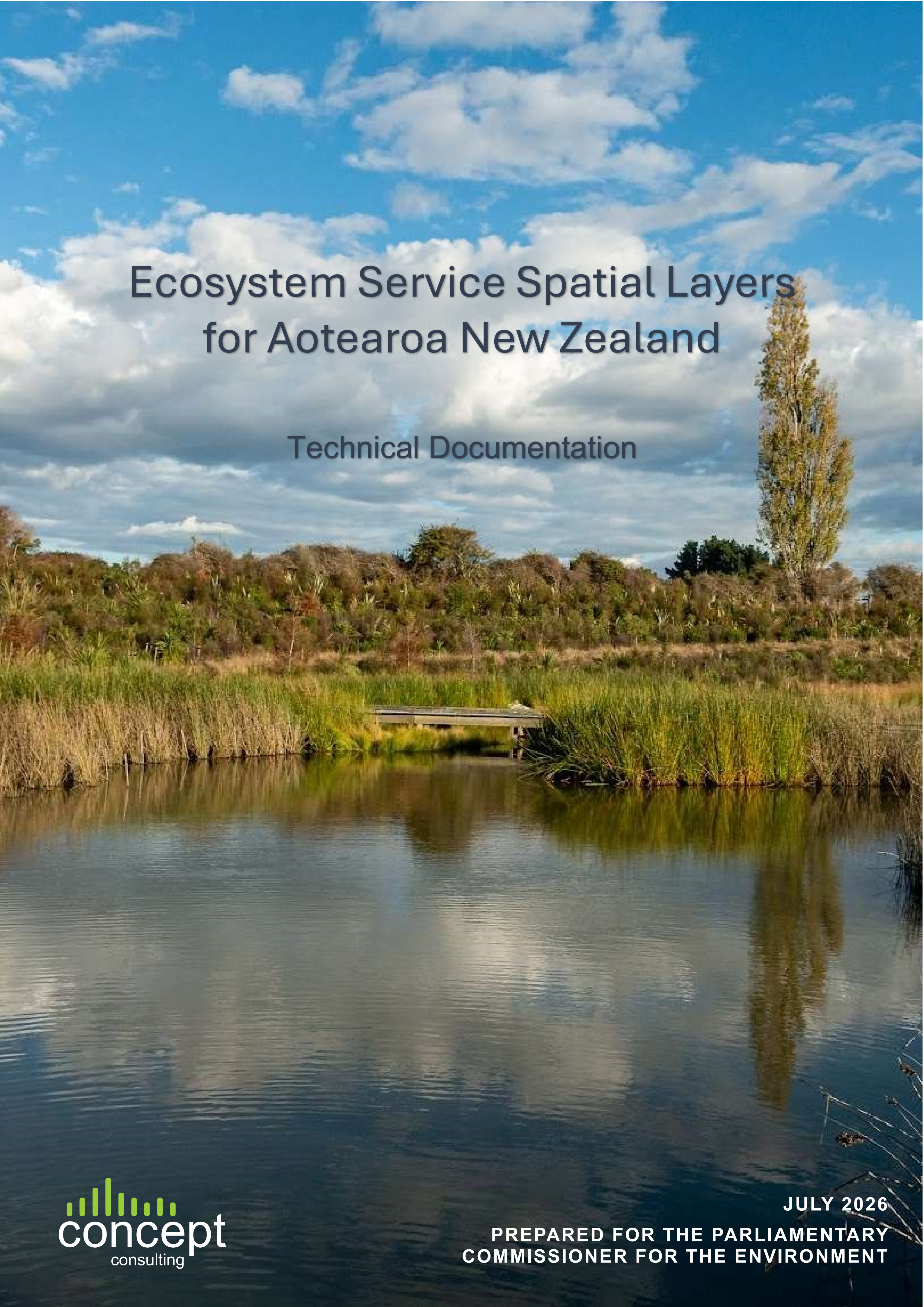

Ecosystem Service Spatial Layers for Aotearoa New Zealand

Technical Documentation

concept consulting

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PREPARED FOR THE PARLIAMENTARY COMMISSIONER FOR THE ENVIRONMENT

Authorship

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Contents

1. Purpose and scope

This report documents the spatial ecosystem service layers developed as part of a project funded by the Parliamentary Commissioner for the Environment (PCE) to support more consistent use of ecosystem service information in New Zealand policy analysis and economic appraisal. It is intended as a technical companion to the guidance document, which explains the broader conceptual framework, valuation principles, policy applications, and caveats for using ecosystem service values.

The purpose of this layer documentation is narrower. It describes what has been mapped, the datasets used, the modelling assumptions applied, and the methods used to convert biophysical indicators into spatially explicit ecosystem service values. For each layer, the documentation sets out the relevant ecosystem service, input data, processing steps, spatial assumptions, valuation approach, uncertainty treatment, and key limitations.

The report is provided for transparency and traceability rather than as a step-by-step technical manual. It explains what was done, why particular methods and assumptions were adopted, and how the resulting layers were generated. Readers with appropriate technical skills should be able to understand, inspect, and adapt the workflow, but the report is not intended to serve as a comprehensive tutorial on ecosystem service modelling.

The layers have been developed primarily around CICES 5.2. The framework distinguishes between ecosystem service potential, realised service flows, and societal benefits, recognising that ecosystem capacity does not always translate directly into benefits received by people. Where relevant, the modelling therefore incorporates spatial relationships between ecosystems and beneficiaries, including hydrological connectivity, accessibility, and distance-decay effects.

Ecosystem service layers necessarily involve judgement, simplification, and uncertainty, particularly where ecological processes, spatial data, and monetary values must be combined. The results should therefore be interpreted as structured, policy-relevant estimates rather than precise measures of value. The companion guidance document should be read alongside this report when applying the layers in cost-benefit analysis, policy assessment, or other decision-support contexts.

2. Intended audience

This technical report is intended for users who require a detailed understanding of how the ecosystem service layers were developed. This includes researchers, environmental modellers, GIS analysts, economists, consultants, government analysts, and other practitioners seeking to understand the data sources, assumptions, processing steps, valuation methods, and limitations underlying the layers.

The report is also intended to support future maintenance, refinement, and extension of the ecosystem service layers. Users wishing to update individual layers, incorporate improved datasets, revise valuation assumptions, reproduce the modelling workflow, or develop additional ecosystem service indicators should find sufficient information to understand the rationale and implementation of the current approach.

Unlike the accompanying guidance document, which focuses on interpretation and application of the results, this report focuses on the technical derivation of the layers. It documents the datasets used, spatial processing methods, modelling assumptions, valuation approaches, uncertainty considerations, and key implementation decisions for each indicator.

The report assumes a degree of familiarity with geographic information systems (GIS), raster and vector spatial data processing, ecosystem service assessment concepts, and environmental valuation methods. Descriptions of spatial processing steps are provided at a level appropriate for readers familiar with common GIS operations rather than as step-by-step software tutorials. Similarly, where Python code has been used, the discussion assumes that readers are capable of inspecting, interpreting, and executing the code if they wish to reproduce or modify the analysis.

3. Frameworks

The project adopts a modern ecosystem service framework that explicitly distinguishes between ecosystem service potential, realised flows, and societal benefits. Ecosystem service potential supply refers to the capacity of ecosystems to generate services under given ecological conditions, while realised flows represent the portion of that capacity that actually reaches and benefits people. This distinction avoids conflating long-term stocks with annual service delivery and follows the conceptual separation of supply, flow and demand set out in the ecosystem services literature. By modelling both the biophysical capacity of ecosystems and the pathways through which benefits accrue to beneficiaries, the approach supports defensible valuation consistent with economic appraisal requirements.

All services are classified using the Common International Classification of Ecosystem Services (CICES) to ensure terminological clarity, comparability, and policy credibility. This classification is intended to support ecosystem accounting as part of the System of Environmental Economic Accounting (SEEA). Although not officially adopted, the SEEA Ecosystem Accounting reference list of ecosystem services is closely aligned. The CICES has been used in Europe, and embedded within a spatial framework aligned with the Mapping and Assessment of Ecosystems and their Services (MAES) initiative from the EU and is designed to operate at MAES Tier 2. This tier involves spatially explicit quantification of ecosystem condition and service supply using regionally or nationally consistent datasets, moving beyond simple land cover proxies while remaining feasible with available data (Burkhard & Maes, 2017; Maes et al., 2016).

Spatial layers are harmonised to a common resolution and projection to enable consistent mapping, aggregation, and scenario analysis.

The project builds on the EU experience, and integrates the best available environmental, socio-economic, and valuation datasets. Environmental layers are drawn from nationally consistent sources, including standardised spatial datasets suitable for macro-scale modelling. Socio-economic layers represent population distribution, infrastructure, and land use as proxies for demand. “Where services involve spatial transmission between ecosystems and beneficiaries, relevant routing or decay models are applied, including hydrological connectivity, distance-decay functions, and service-path attribution approaches. The service-path attribution approach was developed as part of the ARIES modelling framework and is consistent with spatial ecosystem service flow frameworks that distinguish between service supply, demand, realised flows, and beneficiaries (Bagstad et al., 2013; Chalkiadakis et al., 2022; Li & Wang, 2023) .This ensures that mapped values reflect actual service delivery rather than theoretical in situ capacity.

3.1. Monetisation

The monetisation component is grounded in welfare economics and standard cost–benefit analysis (CBA) principles, where changes in social welfare are measured as changes in consumer and producer surplus. The objective is not to price nature in some abstract sense, but to estimate the marginal change in welfare associated with a change in ecosystem service provision under alternative policy scenarios. Values are therefore tied explicitly to defined policy counterfactuals and reflect incremental changes rather than total ecosystem asset values.

For market-related impacts, such as changes in agricultural production, land use, compliance costs, or farm profitability, observed market prices are used to estimate changes in producer surplus. Where environmental policy affects output, input use, or costs of production, these impacts are valued using standard partial equilibrium logic, drawing on farm-level or sectoral modelling consistent with applied CBA practice. Market prices are preferred where they reflect competitive conditions and marginal opportunity costs, as they provide the most direct and least assumption-laden measure of welfare change.

For non-market outcomes, such as improvements in water quality, biodiversity, or recreational amenity, values are based on revealed or stated willingness-to-pay (WTP) measures where robust evidence exists. Revealed preference methods such as travel cost models and hedonic pricing capture use values through observed behaviour , while stated preference methods such as contingent valuation and choice experiments elicit WTP for both use and non-use values under defined scenarios (Hanemann, 1994; Johnston et al., 2017). Where New Zealand studies and reviews exist, these are used as the primary source of values, drawing on the established literature on freshwater recreation and amenity. Where domestic estimates are not available, carefully screened international studies are considered, with preference given to meta-analyses or function transfer approaches that account for income, environmental baseline, and site characteristics (Plummer, 2009; Rosenberger & Phipps, 2007).

In some cases, policy interventions prevent quantifiable damages, such as flood losses, erosion impacts, or health effects from air pollution. Where these damages are measurable and causally linked to ecosystem service provision, avoided damage costs are used as a proxy for welfare change. This approach is consistent with welfare economics where avoided losses represent a gain relative to the counterfactual (Freeman Iii et al., 2014). However, avoided cost methods are applied cautiously and only where the damage–service linkage is well established.

Where ecosystem services are required to meet a defined regulatory or service standard, and WTP evidence is not available or reliable, the least cost alternative approach is used. In such cases, the value of the ecosystem service is proxied by the cost of the next-best engineered or management substitute capable of delivering the same service level. This method is appropriate where the policy objective is compliance with a mandated standard rather than estimation of aggregate WTP, and where the substitute provides an equivalent service outcome.

To summarise, the valuation methods applied to ensure consistency and minimise double counting is:

- Market prices for changes in production, costs, and profits (producer surplus).
- Revealed or stated WTP for non-market outcomes where robust estimates exist.
- Avoided damage costs where mitigation prevents identifiable and quantifiable harm.
- Least cost alternative where benefits reflect delivery of a required service level.

Where primary valuation studies are unavailable or impractical, benefit transfer is used to adapt existing valuation evidence to the policy context under consideration. Benefit transfer involves applying monetary values, valuation functions, or meta-analytic relationships derived from previous studies to a new site or population, with appropriate adjustment for factors such as income, environmental conditions, population characteristics, and policy context. It can be applied across a range of valuation approaches, including willingness-to-pay estimates, avoided damage costs, replacement or least-cost alternatives, and other welfare-consistent valuation methods. Function transfer and meta-analytic transfer are generally preferred over simple unit value transfer because they better account for differences between study and policy sites. While benefit transfer inevitably introduces additional uncertainty, it is widely accepted in applied cost-benefit analysis and ecosystem service assessment where the cost of primary valuation is disproportionate to the decision being informed (NCAVES & MAIA, 2022).

Finally, it is recognised that many ecosystem service values are context-dependent, sensitive to baseline conditions, and subject to substantial uncertainty. Stated preference values depend on scenario design and framing; revealed preference estimates depend on behavioural assumptions; avoided cost methods rely on damage functions that may be imperfectly known. For this reason, monetised values are interpreted as guides to the order of magnitude of trade-offs rather than precise point estimates. They inform the scale and direction of policy choices, not a promise of accuracy to the second decimal place. Sensitivity analysis and clear reporting of uncertainty are therefore integral to the approach.

3.2. Uncertainty

The layer-specific treatment of uncertainty adopted here is consistent with best practice in ecosystem service assessment and applied cost-benefit analysis. In ecosystem service modelling, uncertainty commonly arises from heterogeneous spatial datasets, structural modelling assumptions, and valuation inputs that lack consistent error estimates (Hamel & Bryant, 2017; Seppelt et al., 2011; Walther et al., 2025). Because biophysical and spatial ecosystem-service models often rely on incomplete data, classification choices, modelling assumptions, and error-prone spatial inputs, uncertainty assessment should include transparent documentation of data sources, model structure, assumptions, and known limitations.

Where service flows are modelled, uncertainty stems from routing assumptions, supply-demand coupling, and simplifications of ecological processes; explicit reporting of model structure and parameters is recognised as essential to maintaining credibility and policy relevance (Bagstad et al., 2013).

In valuation, the use of low and high monetary estimates for sensitivity testing is consistent with welfare economics and CBA guidance, which emphasises testing robustness to parameter uncertainty rather than presenting false precision (Freeman Iii et al., 2014) . For stated and revealed preference values, the use of confidence intervals or bounded estimates reflects well-recognised issues of contextual dependence, transfer error, and survey bias (Johnston et al., 2015). Accordingly, monetised results are reported as indicative ranges rather than single point estimates, consistent with ecosystem services scholarship that prioritises decision-relevant magnitude over spurious numerical exactness (Hamel & Bryant, 2017; Seppelt et al., 2011).

At the aggregation stage, the separation of intermediate and final services to avoid double counting follows established ecosystem service accounting guidance .

3.3. Distance decay functions

Distance decay in use values is implemented by treating beneficiaries as origin points and applying an exponential kernel that decreases with distance. The beneficiary raster was convolved with this kernel to create a continuous surface showing where service use is most likely, while keeping total demand unchanged. Following the ES field approach, each kernel was defined by four simple parameters: intensity (maximum use at the origin), range (how far use extends), kernel form (the shape of the decay), and overlap response (how use fields combine when they overlap) (Morris et al., 2024).

4. Overview of layers

Table 1 lists each monetised indicator, its CICES v5 code, a description of the service, the ecosystem features that supply it, and three quality indicators: conceptual validity, data quality, and supply-demand linkage.

Conceptual validity describes how closely the indicator represents the ecosystem service. Direct means the indicator measures the service flow or stock itself. Indirect means it measures a closely related biophysical process or service capacity. Proxy means it uses a substitute variable that is conceptually related to the service but does not directly measure it.

Supply-demand linkage describes how the realised flow is estimated from supply and demand. For most market goods, mapped demand is not required because market prices already reflect demand through observed transactions, and production is assumed to enter wider regional, national, or international markets rather than being consumed only near the point of supply.

For services valued using avoided mitigation cost, such as nutrient regulation and water purification, no explicit supply-demand linkage is modelled. The implicit assumption is that current water quality is below the level demanded by society, so additional nutrient or contaminant removal has value wherever it occurs. Ideally, water quality services would be linked to downstream beneficiaries through hydrological routing and exposure modelling, but that is beyond the scope of this assessment.

The following Table 2 summarises the valuation methods, unit values, and principal source studies and datasets used to develop the ecosystem service layers. Some values shown are weighted averages of more disaggregated estimates, such as values by species, activity, or location. Later sections explain each layer in more detail.

Table 1: Characteristics of the ecosystem service indicators monetised including CICES classification, service suppliers, conceptual validity, data quality, and supply-demand linkage assumptions.

Ecosystem service name	CICES v5	Service	Supplied by	Conceptual validity	Data quality	Supply-demand linkage
Regulating services						
Air quality regulation	2.2.6.1	Capturing/filtering of dust, chemicals and gases from air.	Tree canopy cover	Indirect. Based on vegetation characteristics that influence pollutant removal rather than measured pollutant reductions.	Medium: NZ PM data and damage costs, but tree-cover layer is dated.	Statistical Area 1
Global climate regulation - blue carbon	2.2.6.1	Capturing and storing of carbon by coastal ecosystems	Seagrass, coastal wetlands, and mangroves	Direct. Measures carbon storage or sequestration, which is the service itself.	Medium: NZ habitat coverage, but carbon factors mixed local plus literature.	Global
Coastal storm surge regulation	2.2.1.2	Reducing flooding and erosion from waves	Coastal wetlands and mangroves	Indirect. Estimates hazard attenuation through ecosystems rather than directly measuring avoided damages or wave reduction.	Low: NZ national flood and property data, but international attenuation estimates.	Within same hazard zone
Erosion regulation	2.2.1.1	Soil retention and the ability to prevent and mitigated soil erosion and landslides	Vegetation cover on erosion-prone land and flow paths	Direct. Measures sediment or soil retained that would otherwise be lost through erosion.	High: NZ national erosion modelling, but with simplified assumptions about the contribution of vegetation.	In-situ
Global climate regulation – forest carbon	2.2.6.1	Capturing and storing of carbon by woody biomass	Woody vegetation	Direct. Measures carbon storage or sequestration provided by ecosystems.	High: NZ LUCAS/GHG inventory data.	Global
Local climate regulation	2.2.6.2	Mitigating urban heat and reducing summer cooling demand.	Tree canopy cover	Indirect. Represents cooling potential through vegetation cover and related processes rather than actual temperature reductions experienced by people.	Medium: NZ data for impermeable layer, but dated canopy data.	Statistical Area 1
Fluvial and pluvial flood regulation	2.2.1.3	Mitigation of damage from river and surface flooding	Flood retention volume (InVEST model output)	Indirect. Measures water retention or flood-mitigation capacity rather than observed reductions in flood damages.	Low: NZ asset and flood risk data, but attenuation from international literature.	Sea-draining catchment
Nutrient regulation	2.2.5.1	Reducing nutrient losses to waterways through uptake, retention, and filtering	Nutrient retention capacity of the landscape	Direct. Measures nutrients retained or removed before reaching waterways.	Medium: NZ inputs, but nutrient-load data have age/spatial mismatch issues.	Not linked to demand

Pollination	2.2.2.1	Wild insects contributing to pollen transfer and reproduction of plants.	Potential habitats for wild pollinators	Indirect. Represents pollinator habitat and service potential rather than realised pollination of crops.	Medium: NZ land-cover/ecosystem data, but ecological relationships inferred.	Local dispersal
Regulation of waste	2.2.4.2	Ecosystem ability to filter and decompose organic material in soils.	Soil capacity to decompose organic wastes, proxied by soil drainage and moisture conditions	Proxy. Uses ecosystem characteristics associated with waste assimilation rather than direct measurement of waste treatment.	Medium: based on NZ soil, land use, and livestock data, but decomposition relationships are inferred.	In-situ
Water regulation	2.2.1.3	Slowing, storing and releasing water for later use	Landscape capacity to retain water	Indirect. Measures ecosystem capacity to store and release water rather than actual flow regulation outcomes.	Medium: NZ water-yield and irrigation data, but catchment-scale allocation.	Sea-draining catchment
Water purification	2.2.5.1	Sediment capture before it reaches waterways	Vegetation, soils, wetlands, and riparian areas that slow runoff and trap sediment.	Indirect. Represents the capacity to remove contaminants rather than observed improvements in water quality.	Medium: NZ sediment-retention data, but only covers sediment.	Not linked to demand
Water treatment in wetlands	2.2.5.1	In-situ removal of nutrients, pathogens and sediment by wetlands	Wetlands that intercept stream flows	Indirect. Measures wetland capacity to remove contaminants rather than directly measuring treatment outcomes.	Medium: NZ wetland extent data, but treatment capacity is inferred from 60% of constructed wetland performance.	Not linked to demand
Provisioning services						
Aquaculture	1.1.4.1	Environmental conditions that support aquaculture production.	Coastal habitats suitable for aquaculture species	Direct. Measures production of cultivated aquatic organisms.	High: NZ production/industry data, spatial coverage generally good.	Global market
Biochemical	1.1.5.2	Provision of mānuka-derived compounds used in medicinal, cosmetic, and related products.	Mānuka-dominated vegetation cover	Proxy. Typically based on ecosystem extent or species richness rather than actual harvest of biochemical resources.	Low: limited NZ-specific use data.	Global market
Crops	1.1.1.1	Environmental conditions that support the growth of cultivated plants for human consumption.	Landscape crop yield potential	Direct. Measures agricultural production from ecosystems.	High: NZ agricultural production/land-use data.	Global market
Firewood	1.1.1.3	Woody biomass for household energy use.	Woody biomass in non-conservation areas	Direct. Measures harvestable biomass used for fuel.	Medium: NZ demand data, but harvest location inferred.	Population distance decay
Fodder	1.1.1.1	Environmental conditions that support pasture growth for grazing livestock.	Pasture productivity	Direct. Measures forage production available for livestock.	High: NZ pasture/land-use production data.	Global market

Wild seafood	1.1.6.1	Provision of wild-caught fish, shellfish, and aquatic plants for commercial harvest.	Commercial species presence and abundance by habitat	Direct. Measures harvested wild marine food resources.	High: NZ fisheries data, though spatial allocation is inferred.	Global market
Timber	1.1.1.2	Provision of wood for construction, processing, and industrial use.	Forest growth potential, represented by pine 300-index	Direct. Measures harvestable wood production.	High: NZ forestry production/yield data.	Global market
Freshwater	4.2.1	Provision of freshwater for drinking, domestic, industrial, and agriculture.	Catchment water yield adjusted for land cover	Direct. Measures water supplied from catchments and aquifers.	High: NZ national water-yield data.	Sea-draining catchment
Cultural services						
Recreation - hunting	3.1.1.1	Opportunities for recreational hunting and associated outdoor experiences.	Land cover and habitats supporting hunted species	Direct. Based on recreation values derived from participation and willingness-to-pay studies; spatial allocation is indirect.	Medium: NZ survey/value evidence, allocation inferred.	Population distance decay
Recreation - coastal	3.1.1.1	Opportunities for coastal recreation including swimming, walking, and beach use.	Accessible coastline adjacent to sandy beaches, urban areas, or parkland	Direct. Based on recreation values from observed or stated recreational use; spatial allocation is indirect.	Medium: NZ survey/value evidence, allocation inferred.	Population distance decay
Recreation - freshwater fishing	3.1.1.1	Opportunities for recreational freshwater fishing.	Lakes, rivers, and streams supporting fish populations	Direct. Based on recreation values estimated from angler behaviour and survey evidence; spatial allocation is indirect.	Medium: Mixed NZ and international values; allocation inferred.	Population distance decay
Recreation - freshwater other	3.1.1.1	Opportunities for freshwater recreation other than fishing. Includes swimming, kayaking, and general water-based activities.	Rivers and lakes with high swimmability and water clarity	Direct. Based on recreation valuation studies; spatial allocation is indirect.	Medium: NZ recreation evidence, allocation inferred.	Population distance decay
Recreation - marine boating	3.1.1.1	Opportunities for recreational boating and marine water-based activities.	Accessible near-shore marine environments	Direct. Based on recreation values from user behaviour and survey evidence; spatial allocation is indirect.	High: NZ recreation evidence, allocation mixture of observed and inferred.	Population distance decay
Recreation - mountain biking	1.1.6.1	Opportunities for recreational mountain biking and associated outdoor experiences.	Forest and reserve land suitable for mountain biking access and trails	Direct. Based on recreation values derived from participation and recreation demand studies; spatial allocation is indirect.	Medium: NZ recreation evidence, allocation inferred.	Population distance decay
Recreation – short walk	1.1.6.1	Opportunities for casual nature-based recreation near urban populations.	Urban parks, forests, and reserve land near population centres	Direct. Based on recreation values from use and willingness-to-pay evidence; spatial allocation is indirect.	Medium: NZ participation/value evidence, allocation inferred.	Population distance decay

Recreation - tramping	1.1.6.1	Opportunities for tramping and extended backcountry recreation.	Natural and protected landscapes accessible for tramping	Direct. Based on recreation values derived from participation and willingness-to-pay studies; spatial allocation is indirect.	Medium: NZ recreation evidence, allocation inferred.	Population distance decay
Landscape aesthetics & inspiration	3.1.1.1	Visual amenity, inspiration, and sense of beauty derived from landscapes and ecosystems.	Natural land cover, terrain diversity, and the presence of water features	Proxy. Uses landscape characteristics associated with perceived scenic value rather than direct measurement of aesthetic benefits.	Low: international value evidence adjusted for local exposure hours.	Visual exposure
Terrestrial biodiversity	3.2.2.1	Existence and bequest values associated with terrestrial biodiversity and ecosystems.	Indigenous terrestrial ecosystems and species diversity	Proxy. Uses biodiversity indicators, habitat condition, or species richness as a proxy for existence and bequest values.	Low: international value evidence, single NZ study as upper bound.	Nationwide
Marine biodiversity	3.2.2.1	Existence and bequest values associated with marine biodiversity and ecosystems.	Marine ecosystems and species diversity	Proxy. Uses biodiversity indicators or habitat condition as a proxy for existence and bequest values.	Low: international value evidence	Nationwide

Table 2: Ecosystem service valuation methods, unit values, and principal source studies and datasets used in the development of the ecosystem service layers

Ecosystem service name	Approach	Key sources	Central	Low	High	Unit	Derivation of uncertainty range
Global climate regulation – forest carbon	Shadow price	NZ Treasury (2025)	106	76	135	$/tCO_2e$	From source
Local climate regulation	Avoided energy cost	Ravazdezh & Rivers (2025), NZ variable electricity price	0.3	0.24	0.46	$/kWh	Scaled by Ravadezh effectiveness range
Air quality regulation	Avoided health cost	Ministry of Transport (2023)	639	250	999	$/kg PM10 urban	Scaled by range Hirabayashi 2022
			49	19	76	$/kg PM10 rural	
Water regulation	Least-cost alternative	Irrigation NZ	0.57	0.3	1.12	$/m^3/year	From source
Water purification	Avoided mitigation cost	Fernandez and Daigneault (2017) Matthews, Craggs, Tanner and Holland (2024, 2026)	40	33	110	$/t sediment	From typology ranges in Matthews et al. 2024
Nutrient regulation	Avoided mitigation cost	Abatement curves from Matthews, Craggs, Tanner and Holland (2024, 2026)	N/A	21	226	$/kg N or P	From typology ranges in Matthews et al. 2024
Erosion regulation	Avoided productivity loss	Soliman (2005)	0.12-5.04	0.07-4.20	1.45-8.91	$/t soil for sheep and beef	Cost for adjacent farm type
Fluvial and pluvial flood regulation	Avoided damage cost	OECD (2020)	20%	10%	30%	of residual risk	From source
Coastal storm surge regulation	Avoided damage cost	Narayan et al. 2017, Barbier et al. 2013, Fairchild et al. 2021	5%	10%	20%	Damage reduction	Range from Fairchild studies
Pollination	Replacement cost	Goodwin 2012; MPI Apiculture Monitoring	200	150	300	$/hive	From Apiculture Monitor
Regulation of waste	Avoided treatment cost	Council trade-waste charges	0.8	0.5	1.2	$/kg manure BOD	From source
Global climate regulation – blue carbon	Shadow price	NZ Treasury CBAx carbon price	106	76	135	$/tCO_2e$	From source
Water treatment in wetlands	Avoided mitigation cost	Daigneault et al. 2017	27	14	54	$/kg N	From source
	Avoided mitigation cost	Daigneault et al. 2017	136	68	272	$/kg P	From source
	Avoided mitigation cost	Daigneault et al. 2017	4.1	2.0	8.2	$/t sediment	From source

	WTP for pathogen reductions	Tait 2016	69	58	80	$/ha for band A	From source
Crops	Simple factor share	Fresh Facts 2025, MPI Vineyard monitoring, AIMI 2025	0.31	0.13	0.50	Average contribution proportion	From JRC 2024
Fodder	Simple factor share	DairyNZ 2025	0.39	0.2	0.5	$/kg DM	Plausible range
	Production value	Beef + Lamb 2025	100%	58%	145%	of operating profit	5-year profit variability
Timber	Market price	Manley 2021	40.17	27.67	61.75	$/unit 300-index	From source
Firewood	Replacement cost	Cost of alternative heating by heat pump ($0.30/KWh, COP = 3)	28	15	39	$/GJ	Derived from variation in COP
Wild seafood	Market price	Fisheries catch, Fishserve port prices	100%	69%	125%	of port price/kg	Derived from marine account
Aquaculture	Simple factor share	MPI Aquaculture Performance (2024), Fishserve	60%	40%	80%	of production value	Plausible range
Freshwater	Productivity proxy	Irrigation NZ, BRANZ 2018	0.02 -1.00	0.019-0.5	0.021- 1.5	$/m3	Plausible range
Biochemical	Proxy market value	MPI Apiculture Monitoring	330	264	402.6	$ million (total)	Derived from market variability
Recreation - hunting	Stated and revealed preference	Kerr & Woods 2010, Kerr and Abell 2014, Kerr 2019, Gren & Kerr 2022	$259	$187	$328	Weighted avg $/trip	From sources
Recreation – short walk	Composite	NZTA 2013	5.14	4.11	7.41	$/trip	From source
Recreation - tramping	Revealed preference	Kaval and Yao (2010) day tramp	198.49	23.82	373.17	$/trip	From source
	Revealed preference	Kaval and Yao (2010) overnight	793.97	95.28	1492.67	$/trip	From source
Recreation - mountain biking	Revealed preference	Dhakal et al. (2012)	67.00	51.00	106.00	$/trip	From source
Recreation – coastal	Stated and revealed preference	Matthews et al. (2018)	47.21	21.21	131.35	$/trip	From source
Recreation freshwater fishing	Stated and revealed preference	Kerr (2003), Jiang (2015)	120.00	78.00	605.00	$/trip	Cross-study range
Recreation – freshwater other	Stated and revealed preference	Kaval and Yao (2007)	115.73	55.42	154.85	$/trip	From source
Recreation - marine boating	Stated and revealed preference	Kerr and Latham (2011)	270.00	202.50	339.00	$/trip	From source
Aesthetic	Stated and revealed preference	Brander & Koetse, (2011)	1.37	0.99	11.68	$/exposure hour	From source
Terrestrial biodiversity	Stated preference	Subroy et al. (2019) - international meta-analysis	46	4	85	$/household	From source

Marine biodiversity	Stated preference	Chhun et al. (2013), Rojas-Nazar (2022)	22	7	54	$/household	From sources

5. Regulating services

5.1. Global climate regulation – forest carbon

Explanation:

Carbon stocks were used to represent ecosystem service supply because they reflect the standing pool of carbon stored in vegetation and therefore the long-term capacity of ecosystems to regulate climate under a given land cover.

Carbon sequestration was used to represent ecosystem service flow because it captures the annual uptake of carbon from the atmosphere and thus the realised climate regulation delivered in a given year. Stocks and flows are related but not interchangeable: ecosystems with high carbon stocks may have low sequestration rates, while rapidly growing systems may deliver high sequestration despite low stocks. Using carbon stocks as supply and sequestration as flow follows established ecosystem services frameworks and avoids conflating long-term storage capacity with short-term carbon uptake.

LUCAS is used as the spatial unit of analysis because it aligns closely with national inventory reporting and provides consistent soil and biomass carbon factors by land-use class. The LUCAS framework distinguishes between pre- and post-1990 forestry and between exotic and natural forest, with sub-classes that differentiate regenerating versus tall forest and pine versus other softwoods. Although carbon stocks have been estimated for more species-specific forest classes in LCDB (Paul et al., 2021; Paul & Wakelin, 2023) , full alignment would add unnecessary complexity.

No spatial layer is available that represents forest age beyond the pre- and post-1990 distinction, so carbon stocks for post-1989 forest are based on average age assumptions consistent with National Exotic Forest Description (NEFD) tables. Sequestration is estimated using target accounting removals for 2023, averaged over each hectare in each of the relevant LUCAS classes.

Monetisation

Valuation is based on the New Zealand Treasury's target-consistent shadow price of carbon, as specified in the Treasury CBAx guidance and tool. The current estimate for 2026 is NZ$106 per tonne of CO_2-equivalent (New Zealand Treasury, 2025) for the central pathway, $76 for low, and $135 for high.

Data sources:

- LUCAS NZ Land Use Map 2020 v005
- Soil carbon from 2025 GHG Inventory table A5.2.9
- Biomass carbon from 2025 GHG Inventory tables 6.6.4, 6.5.6, 6.7.3
- ETS yield tables

Limitations

Given the limited spatial extent of peat soils relative to the mapped land-use classes, soil carbon was assumed to reflect mineral soils, consistent with MAES-style ecosystem service mapping.

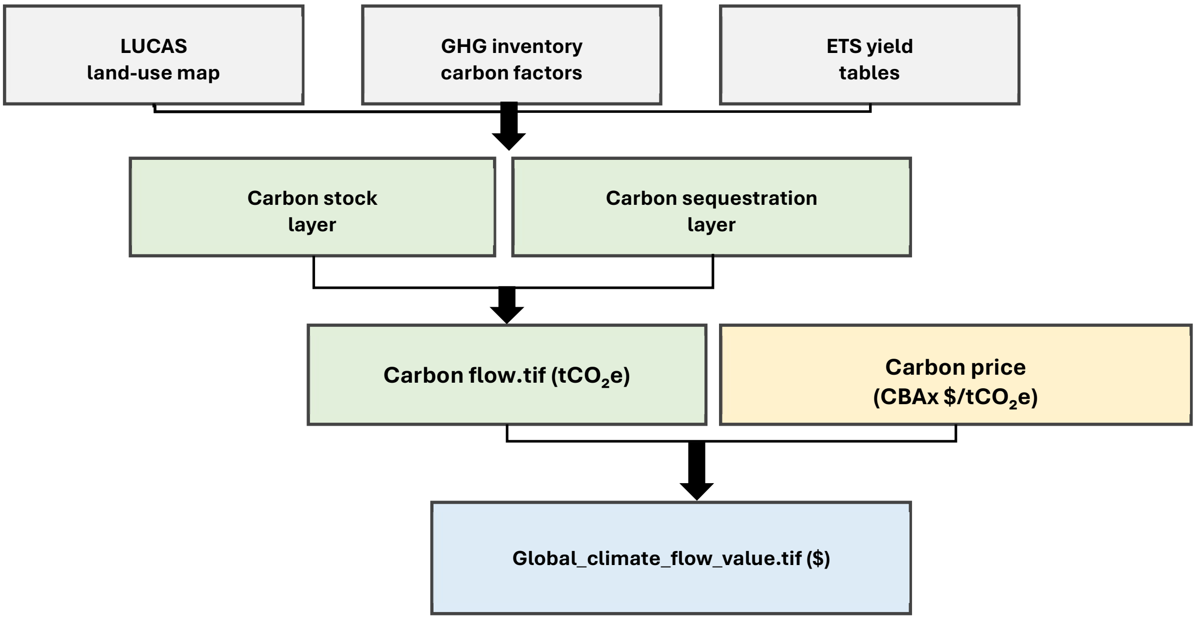

Figure 1: Flow diagram of inputs/outputs for global climate regulation

Potential future improvements:

Future versions of the global climate regulation layer could be improved by incorporating more detailed information on forest age, management regime, and harvesting cycle. In particular, distinguishing between recently planted, maturing, mature, harvested, and permanent forest would allow sequestration flows to be estimated with greater spatial precision.

The layer would also benefit from clearer separation of natural and anthropogenic ecosystems using the national ecosystem typology developed by Ministry for the Environment (Sprague & Wiser, 2024) and data on natural ecosystems for carbon stocks and potential sequestration (Ausseil et al, 2013). Plantation forestry should be distinguished from natural forest because it is a modified ecosystem, is managed primarily for production, and may be treated differently in ecosystem accounting and international reporting. In particular, the Convention on Biological Diversity specified reporting on ecosystem services should focus on the contribution of natural capital instead of human produced capital[1]. This distinction would also help avoid conflating long-term carbon storage in natural ecosystems with shorter-term carbon sequestration associated with production forestry.

Steps:

- Compiled a table of carbon stocks and sequestration flows for each LUCID and SUBID class.
- Join on LUCAS features
- Rasterize both carbon stock and carbon flow.
- Multiply by 44/12 to convert to tCO2-e and save as global_climate_flow.tif
- Multiply by $106, $76, and $126 to create the central, low and high rasters.

[1] https://www.gbf-indicators.org/metadata/headline/11-1

5.2. Local climate regulation

Explanation:

Local climate regulation is the ecosystem service by which vegetation, especially tree canopy, moderates local temperatures through shading, evapotranspiration, and the reduction of heat stored in built surfaces. In urban and peri-urban environments this service reduces the intensity of heat accumulation associated with impervious surfaces, helping to offset urban heat effects and improve thermal comfort.

The supply of local climate regulation is represented by tree canopy cover, derived from the tree cover raster (Hansen et al., 2013) as an indicator of shading and evapotranspiration capacity. This measure is applied across the entire country and reflects potential supply, while the demand-side factors mean that the realised flow is effectively negligible in rural areas.

Demand for the service is represented by impervious surface coverage at the Statistical Area 1 (SA1) level, adjusted by local temperature conditions to reflect that the cooling benefit of canopy is greater in warmer areas where heat accumulation and thermal stress are higher. Temperature adjustment was based on annual cooling degree days (CDD), calculated from WorldClim monthly minimum and maximum temperature surfaces. The WorldClim data was used instead of the NZ environmental data stack (nzenvds) because it provides monthly observations.

Monthly mean temperature was derived as the average of monthly maximum and minimum temperature, and CDD were computed as the positive difference between monthly mean temperature and a base temperature of 18°C, multiplied by the number of days in each month and summed across the year. This provides a spatially explicit indicator of cooling demand intensity, with higher CDD values representing areas where buildings are more likely to require active cooling. Population was not used as a primary exposure variable because SA1 resident counts may not reflect day-time occupancy in commercial and mixed-use areas, where cooling demand can be substantial despite lower resident populations.

Monetisation

Local climate regulation was monetised by estimating the value of cooling energy avoided due to existing urban tree canopy. National residential space-cooling energy use (1,028 GWh) was obtained from the Energy End-Use Database (EEUD 2023) and allocated spatially across 100 m grid cells in proportion to households, cooling degree days and impervious surface area, reflecting the distribution of cooling demand. The cooling reduction attributable to canopy was modelled as a function of tree cover and imperviousness, with a maximum potential reduction of 30% under conditions of 100% canopy cover in fully impervious urban areas, consistent with recent empirical evidence on canopy-related cooling energy reductions (Ravazdezh & Rivers, 2025). The observed cooling energy is treated as the post-canopy baseline, and the counterfactual “no-canopy” energy requirement was derived analytically. The avoided energy attributable to existing canopy was calculated as

$$E_{saved} = E_{obs}\left(\frac{r}{r-1}\right)$$

where r is the estimated fractional reduction. Avoided kilowatt-hours were then valued using an average residential electricity price of $0.30 per kWh to produce a per-cell annual dollar estimate.

Household density was used as a spatial proxy for the distribution of commercial cooling demand, recognising that households account for approximately one third of total space cooling energy in the EEUD dataset and that commercial premises are assumed to co-locate with areas of higher residential intensity.

The low and high sensitivity bounds for local climate regulation were derived from the heterogeneity reported by Ravazdezh and Rivers (2025), who estimated that a 10 percentage-point increase in nearby canopy cover reduced summer residential electricity consumption by between 2.4% and 4.6% depending on tree orientation relative to buildings. Scaling these marginal effects to the theoretical maximum of 100% canopy cover implies an approximate range of 24% to 46%.

Data sources:

- statsnzstatistical-area-1-2023-generalised
- lds-nz-road-centrelines-topo-150k
- lds-nz-building-outlines-FGDB
- Global Forest Watch canopy cover 2000 (https://data.globalforestwatch.org/documents/5fb3275e080e497fa44174d2b14d4b7c)
- WorldClim monthly minimum and maximum temperature surfaces for 2024

Limitations

The canopy cover layer is relatively old but it is still the better option because LCDBv6 is too coarse for urban areas. There are some new subdivisions that are incorrectly showing high values because the removal of tree cover has not been factored in.

Future improvements:

Incorporate the forthcoming Bioeconomy Science Institute (BSI) urban greenspace mapping work for major New Zealand cities, which is expected to distinguish tree canopy from grass cover. This would substantially improve representation of urban cooling effects and local climate regulation services, as tree canopy generally provides stronger shading and evapotranspiration benefits than low vegetation. The current modelling is limited by the availability of nationally consistent urban vegetation layers.

Develop more detailed and nationally consistent maps of impermeable surfaces. Existing datasets do not adequately capture fine-scale urban variation in sealed surfaces such as roads, car parks, roofs, and paved areas, all of which strongly influence urban heat retention and runoff. Local analyses, such as the Dunedin City Council impermeable surfaces mapping project, demonstrate the level of spatial detail that would improve future modelling of urban climate regulation services.

From a primary sector perspective, shelter, forestry blocks and even individual trees can play an important role in both providing shade for livestock when higher temperatures are causing heat stress, and in providing shelter against storm, snow, and cold wind events. Shade is important to maintain a healthy metabolic function in livestock, while shelter can reduce effective wind speeds and is critical during lambing and calving to protect new-born stock. While there may be limited information on these animal welfare and farm management benefits, these benefits could be quantified in future.

Steps

Create a layer of tree cover (ES supply)

- Merge the 3 tree cover tiles that cover NZ
- Reproject to EPSG:2193 and compress
- Save as local_climate_supply.tif

Create a layer to approximate impermeable areas

- Inputs:

  - Roads = D:\Data\LINZ\lds-nz-road-centrelines-topo-150k\ nz-road-centrelines-topo-150k.gpkg|layername= nz_road_centrelines_topo_150k
  - SA1 = D:/Data/Stats NZ geography/statsnzstatistical-area-1-2023-generalised-GPKG/statistical-area-1-2023-generalised.gpkg|layername=statistical_area_1_2023_generalised
  - Buildings = "D:\Data\LINZ\lds-nz-building-outlines-FGDB\ buildings_fixed_geometry.gpkg|layername=fixed_geometry
- Add a calculated field road_width to lds-nz-road-centrelines-topo-150k: road_width = CASE WHEN lane_count = 1 THEN (lane_count * 5 + 2) / 2 ELSE (lane_count * 5 + 4) / 2 END
- Use "Geometry by Expression" with expression "buffer($geometry, "road_width")"
- Intersect with SA1, keeping only the id field SA12023_V1_00.
- Dissolve by SA1
- Save as roads_SA1_dissolved
- Using the buildings layer, join attributes by location to get SA1 for each building.
- Add a calculated field building_area_m2
- Use "Statistics by Categories" to sum building area by SA1
- Join SA1 on SA1_dissolved and building statistics to get the road area and built area
- Add a calculated field: min(("impervious_building_sum" + "impervious_road_area") / "AREA_SQ_KM" / 1000000,1)
- Save layer as: local_climate.gpkg|layername=SA1_impervious"
- Rasterise SA1_impervious

Calculate flow

- Calculate Census 2023 households per hectare and rasterise
- Run calculate_cooling_degree_days.py to calculate base cooling demand
- Run local_climate_flow.py to calculate flow_value and normalised flow

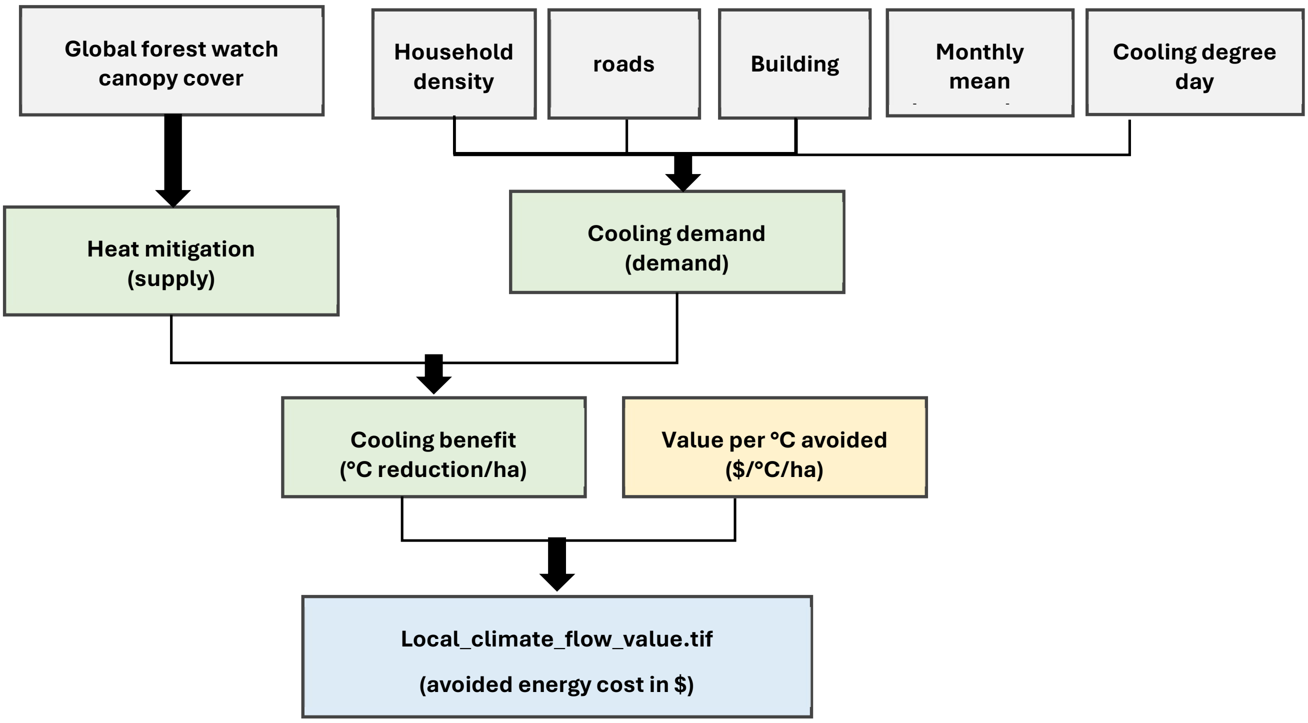

Figure 2: Flow diagram for local climate regulation

5.3. Air Quality regulation

Explanation:

Air quality regulation is the ecosystem service through which vegetation removes airborne particulate matter from the atmosphere via dry deposition to leaf and canopy surfaces. By reducing ambient particulate concentrations and human exposure, tree cover contributes to improved respiratory and cardiovascular health outcomes and avoids associated economic damages. The assessment focuses on PM10 because nationally consistent monitoring data and damage cost values are available for this pollutant in New Zealand, and PM10 encompasses both fine and coarse particulate fractions relevant to health impacts and policy appraisal.

Air quality regulation was modelled as particulate removal by vegetation using a dry deposition approach consistent with the i-Tree Eco methodology (Hirabayashi, 2022). In this framework, annual removal mass is calculated as:

$M = Vd \times C \times A \times t$

where M is annual pollutant removal (kg per year), Vd is effective deposition velocity (m per second), C is ambient concentration (kg per cubic metre), A is area (square metres), and t is time (seconds per year). Deposition velocity per unit leaf area was taken from the i-Tree PM table for a representative near-canopy wind speed of 3 m per second (0.15 cm per second), and scaled by a leaf area index (LAI) of 6 to obtain an effective Vd of 0.009 m per second. For a 1-hectare grid cell (10,000 m²), this yields a removal coefficient of 0.00284 tonnes PM10 per year per microgram per cubic metre of ambient concentration. Annual mean PM10 concentrations were obtained from the NZTA national background air quality dataset (NZTA, 2023), which reports concentrations in micrograms per cubic metre.

Monetisation

Economic value was estimated by multiplying tonnes removed by the NZ transport damage cost per tonne of PM10. Damage costs were taken from the Domestic Transport Costs and Charges Study (Ministry of Transport, 2023, Table 2.2) and inflated to 2026 NZD. The resulting values applied in this analysis are $639 /kg PM10 for urban areas and $49 for rural areas. Urban and rural values were applied using a population density threshold of 4 persons per hectare (400 persons per square kilometre), as per the definition stated in the report. The realised annual ecosystem service value per hectare was therefore calculated as:

Value = DamageCost (urban or rural) × (TreeCover / 100) × C_PM10 × 0.00284

where TreeCover is percent canopy cover per grid cell and C_PM10 is annual mean PM10 concentration (micrograms per cubic metre).

As the NZTA guidance does not provide an uncertainty range for PM_{10} damage cost values, the analysis instead applies the minimum and maximum deposition velocities reported in the i-Tree Eco methodology. These imply scaling factors of approximately 39% (low) and 156% (high) relative to the central estimate.

Table 3: Damage cost values per kg PM10 for central, low and high

Category	Central	Low	High
Urban (>4 residents/ha)	639	250	999
Rural (<4 residents/ha)	49	19	76

Data sources

- NZTA Background-PM-concentrations-by-CAU-May22 (https://nzta.govt.nz/assets/Highways-Information-Portal/Technical-disciplines/Air-quality/Planning-and-assessment/Background-air-quality/Background-PM-concentrations-by-CAU-May22.xlsx)

- Statistics NZ CAU 2013 geography
- statsnz-2023-census-totals-by-topic-for-individuals-by-SA1
- Global Forest Watch canopy cover 2000 (https://data.globalforestwatch.org/documents/5fb3275e080e497fa44174d2b14d4b7c)

Steps

- Join NZTA on CAU_2013 to bring in the attribute PM10 annual
- Rasterise and save as PM10.tif
- Normalise to a range 0 to 100.
- Save as air_quality_demand.tif

Calculate the average treecover for each SA1

- Zonal statistics on treecover
- Save as air_quality_supply.tif

Calculate the flow indicator for each SA1

- Use raster calculator:

```
where(pop_density>4,urban_value,rural_value)*1000*(treecover/100) * PM10_t * 0.00284
```

- Save as Air_quality_flow_value.tif

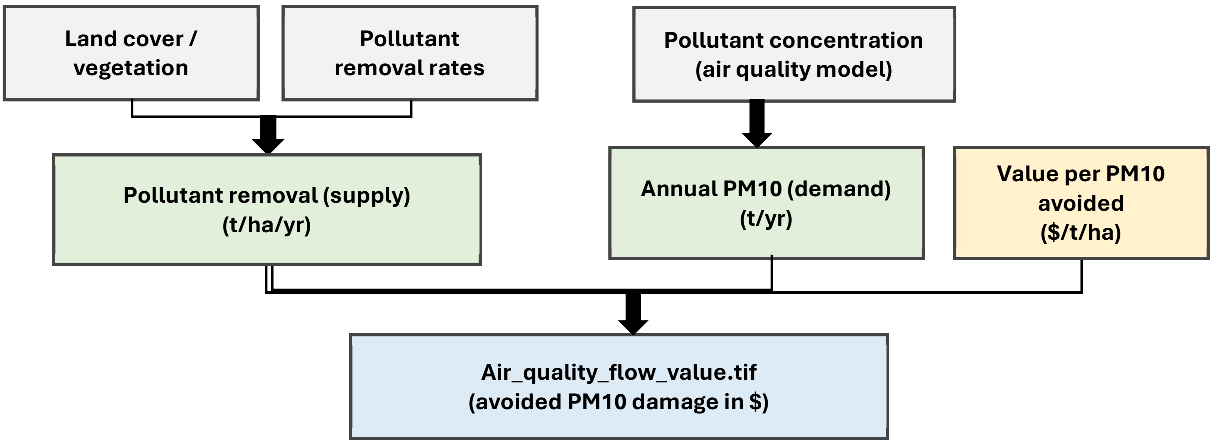

Figure 3: Flow diagram for air quality regulation

Limitations

Demand for air quality regulation was based on residential exposure, implicitly assuming that exposure at home dominates overall long-term exposure, while exposure at workplaces and educational settings was not explicitly modelled. Residential exposure is generally the most important proxy for long-term exposure, especially for children or elderly people (Klepeis et al., 2001).

The use of nationally consistent air pollution damage values derived from NZTA guidance may overstate welfare impacts in remote rural areas where population exposure, baseline pollution concentrations, and associated health risks are comparatively low.

Future improvements

Future improvements could include incorporating forthcoming urban tree-cover data from the national greenspace mapping work, expected in June 2026 for the six major cities. The treatment of exposure could also be refined beyond the current NZTA urban/rural damage-cost thresholds to incorporate population density, demographic vulnerability, and spatial variation in exposed populations. Finally, the indicator could transition to $PM_{2.5}$ if monitoring coverage, spatial modelling, and monetised health-impact estimates are sufficiently developed for consistent national application.

5.4. Water regulation

Explanation

Water regulation refers to the capacity of landscapes to temporarily retain precipitation and release it more gradually, thereby buffering flows and supporting supply reliability during periods of deficit. It differs from water provisioning, which measures the total volume of water yielded and available for abstraction (i.e. runoff or yield). Provisioning concerns the quantity of water supplied, whereas regulation concerns the timing and reliability of that supply.

The supply indicator is baseflow contribution, represented by the B.tif output from the InVEST Seasonal Water Yield model. This raster estimates each cell's contribution to slow-release flow reaching the stream network, after accounting for local recharge and the likelihood that water is evapotranspired before reaching a stream. It is therefore more closely aligned with the water regulation service than a simple retained-water proxy, because it represents delayed contribution to streamflow rather than total water not exported as annual runoff. InVEST reports B.tif in mm/year, but the model documentation recommends interpreting it as a relative indicator rather than a calibrated hydrological volume. For this reason, B.tif is used here as a spatial proxy for each cell's relative contribution to baseflow regulation.

Demand for water regulation was represented by irrigation water use, reflecting the role of ecosystem-mediated retention in supporting supply reliability during seasonal deficits. Municipal takes were not included because urban water supply is usually not dependent on natural storage and baseflows. Because water retention has downstream benefits, the indicator is average irrigated hectares per sea-draining catchment hectares.

Realised flow was calculated at the sea-draining catchment scale to reflect the downstream use of upstream baseflow regulation. Although the InVEST Seasonal Water Yield model provides a spatial estimate of each cell's relative contribution to baseflow, it does not identify which individual cells supply water to specific irrigation takes. Sea-draining catchments were therefore used as the spatial unit for linking ecosystem-mediated baseflow contribution with irrigation demand.

Eco-index sea draining catchments are used in preference to the layer published by Ministry for the Environment, because the Eco-index layer simplifies the smaller catchments that proliferate near the coast.

For each sea-draining catchment, modelling irrigation demand (Aqualinc, 2021) was used to define the total catchment-level value associated with water regulation. This value was then allocated back to source cells in proportion to their B.tif value, so that cells with a larger modelled contribution to slow-release/baseflow received a larger share of the catchment value:

$$V_i = V_{catchmen} \left(\frac{B_i}{\sum_c B_{catchment}} \right)$$

Where V_i is the water regulation value assigned to cell i, V_c is the total irrigation-related water regulation value for catchment c, and B_i is the InVEST baseflow contribution for cell i.

This approach preserves the catchment-level value total while assigning value only to cells that contribute to modelled baseflow. Because B.tif is best interpreted as a relative baseflow contribution indicator rather than a calibrated hydrological volume, the resulting raster should be interpreted as a spatial attribution of irrigation-related water regulation value, not as an observed volume of water delivered to irrigation users.

Monetisation

Realised value is obtained by multiplying realised flow volume by the annualised price of storage capacity. This approach values the reliability or buffering function of ecosystems, rather than the water itself, and therefore avoids double-counting the provisioning service. Value per m3 stored is calculated as:

$$P = CAPEX * CRF + OPEX$$

where CAPEX is the capital cost per cubic metre of storage capacity, CRF is the capital recovery factor, and OPEX represents annual operating and maintenance costs per cubic metre of storage capacity.

Recent estimates indicate that modern dam construction costs range from approximately \$5 to \$20 per m³ of storage capacity (Irrigation NZ, 2023). These values are used as lower and upper bounds, with \$10 per m³ adopted as the central estimate. Annual dam operating costs are assumed to average 0.25% of capital costs (Petheram & McMahon, 2019).

Treasury guidance currently recommends a 2% discount rate for environmental impacts and an 8% rate for commercial investments. A central rate of 5% is adopted here as an intermediate assumption, resulting in a capital recovery factor (CRF) of 0.055 for a 50-year asset life. Under these assumptions, the resulting annualised storage price ranges from approximately \$0.30 to \$1.12 /m³/year, with a central estimate of \$0.57 /m³/year.

Data

- nzenvds-total-annual-precipitation-v10
- nzenvds-precipitation-of-the-driest-month-v10
- nzenvds-precipitation-of-the-driest-quarter-v10
- nzenvds-precipitation-of-the-wettest-month-v10
- nzenvds-precipitation-of-the-wettest-quarter
- nzenvds-rainfall-to-potential-evapotranspiration-ratio-v10
- mean-monthly-wet-days (https://niwa.co.nz/climate-and-weather/mean-monthly-wet-days)
- Global Hydrologic Soil Groups (HYSOGs250m) (https://www.earthdata.nasa.gov/data/catalog/ornl-cloud-global-hydrologic-soil-group-1566-1)
- Eco-index catchments (https://eco-index.koordinates.com/layer/115002-eco-index-catchments-for-nz/)
- Irrigation demand by Freshwater Management Unit

Steps

Prepare and run InVEST Seasonal Water Yield

- Create a biophysical of evapotranspiration coefficients for each LCDB6 class as per InVEST guidance.
- Distribute rainfall and EVT over 12 months using the python script *Derive_monthly_precip_and_PET.py*
- Reproject HYSOGs250m soilgroup and clip to NZ extent
- Assign each site in mean-monthly-wet-days a numeric index and save the data as climate_zone_table.csv

- Use Voronoi polygons to assign each cell to a weather station in mean-monthly-wet-days. Save as climate_zones.tif
- Sum the output B.tif by sea-draining catchment and rasterise as catchment_total_baseflow.tif

Create demand Layer

- Multiply irrigation demand (mm) by 10 to get volume (m^3)
- Multiply by $0.57
- Sum over the entire sea draining catchment
- Rasterise and save as catchment_total_value.tif

Create flow layer

- Do the raster calculation
  “where((Bc>0), Vc*B/Bc,-9999)”
- Save as water_regulation_flow_value.tif

Future improvements

Future improvements should focus on strengthening the hydrological linkage between supply cells and beneficiaries. The current method allocates catchment-level irrigation-related value back to source cells in proportion to their InVEST baseflow contribution, but it does not model the specific pathways by which water from each cell reaches individual abstraction points. A more refined approach would use a routing model that links each raster cell to downstream water takes through the river network, accounting for flow direction, stream connectivity, attenuation, existing abstractions, return flows, storage, and seasonal demand. This would allow water regulation value to be assigned only to cells that plausibly contribute to water available at specific take locations, rather than to all baseflow-contributing cells within the same sea-draining catchment.

Future extensions could also incorporate water-regulation benefits beyond irrigation (and flood protection, which is captured separately). Baseflow and delayed recharge can support the reliability of stock water, run-of-the-river hydropower, recreational, ecological and cultural water uses. Some of these benefits occur through surface-water pathways, where slow-release flow sustains rivers and streams, while others occur through groundwater pathways, where infiltration and recharge support aquifers, springs, shallow groundwater takes and connected surface flows. Future work could incorporate additional beneficiary layers to represent a broader set of water-regulation benefits.

5.5. Water purification

Explanation

Water purification is a regulating ecosystem service whereby ecosystems improve water quality by removing, transforming, or retaining contaminants before they reach downstream environments or human users. These processes include sediment trapping, nutrient uptake, filtration by vegetation and soils, and microbial breakdown of pollutants. In ecosystem accounting frameworks, such services are often represented using measurable biophysical indicators that capture the ability of ecosystems to intercept pollutant flows.

In this analysis, sediment retention is used as a proxy indicator for the water purification service. Spatially explicit data on several purification processes, such as pathogen removal or microbial transformation of contaminants, are not available. Sediment retention nevertheless captures an important component of the service because vegetation and soils that intercept eroded sediment reduce the downstream transport of suspended particles and associated contaminants, thereby improving water quality. Nutrient retention is treated separately as its own indicator.

Supply indicator

A separate supply layer for water purification was not mapped because no defensible spatial data exist to represent sediment trapping capacity independent of sediment loads, and the available models (SDR/NZEEM) provide only realised retention (flow).

Flow indicator

Water purification is represented using NZEEM sediment retained, which measures the amount of mobilised sediment intercepted or trapped before reaching streams. This indicator reflects a filtration and retention process acting on sediment already in transit, benefiting downstream water quality. It does not represent reduced erosion at the source, and therefore does not overlap conceptually with erosion regulation. Using local soil loss avoided for erosion regulation and sediment retained for water purification avoids double counting and aligns each service with its distinct biophysical process.

Demand indicator

Water purification demand is represented by the total sediment load requiring interception (approximated as sediment retained plus sediment lost). It is not weighted by downstream population because this would require assumptions about how far mobilised sediment is transported and where along the river network benefits are realised, which cannot be resolved without an explicit sediment transport and deposition model. In the absence of such a model, population weighting risks attributing demand to beneficiaries who may not be exposed to the sediment load. Demand is therefore represented using the biophysical sediment load requiring interception.

Monetisation

Sediment retention is valued using a least-cost mitigation approach. This reflects the cost that would be incurred to achieve an equivalent reduction in sediment loss if natural retention processes were absent. Under binding freshwater quality constraints, the marginal cost of compliance provides an appropriate proxy for the value of the regulating service.

At present, it is not possible to translate retained sediment into specific downstream welfare benefits. Sediment transport pathways, attenuation processes, and exposure of beneficiaries (e.g., recreation, drinking water abstraction, ecological thresholds) cannot be resolved at sufficient spatial resolution to quantify avoided damages directly. In the absence of a fully specified damage function, it is assumed that any reduction in anthropogenic sediment is beneficial, consistent with freshwater quality objectives and regulatory settings. The mitigation cost approach therefore provides a conservative lower-bound estimate of the service value.

Fernandez and Daigneault (2017) analysed erosion mitigation through targeted land retirement in the Waikato District. They report marginal costs ranging from $12 to $515 per tonne of sediment reduced (approximately $16 to $686 in 2026 prices). These results demonstrate that mitigation costs are highly convex: early reductions can be achieved at low cost by retiring the most erosion-prone land, but marginal costs rise sharply once low-cost opportunities are exhausted. In practice, the cheapest land may already have been retired, meaning marginal costs in many catchments are likely to exceed the lower end of this range.

In a broader approach, Matthews, Craggs, Tanner and Holland (2024; 2026) estimated marginal abatement costs for optimised sediment reduction using bundles of edge-of-field mitigation systems (e.g., riparian planting, detainment bunds, constructed wetlands) across diverse rural landscapes. These studies show that marginal costs depend on landscape characteristics, hydrological pathways, and the level of prior mitigation. Reported marginal costs range from $33 to $110 per tonne. For moderate levels of sediment reduction in landscapes where slope is not too severe, marginal costs generally remain around $40 per tonne. In the case where the slope is too high for interception, the retirement approach and associated costs from Fernandez and Daigneault are presumably applicable.

For the purposes of this analysis, a central estimate of $40 per tonne is adopted, with a sensitivity range of $33 to $110 per tonne. This reflects representative marginal mitigation costs under typical pastoral landscape conditions and avoids reliance on average or damage-based unit values that may not reflect compliance costs.

Data sources

- Iris-sediment-retained (NZEEM)

Steps

- Align the grid of NZEEM sediment retention to other layers (SMAP is the base) and export as water_purification_flow.tif
- Multiply this raster by $40 and export as water_purification_flow_value.tif

Limitations

This assessment represents water purification using sediment retention only, which captures just one dimension of water quality regulation. While sediment is often correlated with contaminants such as *E. coli*, particularly in agricultural catchments, it is at best a crude proxy and does not explicitly represent microbial sources, survival, or in-stream processes. As a result, the water purification service is likely to be under- or mis-represented in settings where microbial contamination is weakly linked to sediment dynamics.

Mapping the supply of water purification for E. coli was not undertaken because it requires explicit representation of microbial sources, transport pathways, die-off, and interception processes, and no suitable InVEST model or comparable spatial tool was available within the scope of this project to support such modelling at the required scale. Catchment-scale microbial models do exist in New Zealand, most notably the CLUES (Catchment Land Use for Environmental Sustainability) framework, which has been used to estimate E. coli loads and attenuation through river networks. However, implementing CLUES requires additional data preparation, calibration, and model setup beyond what was feasible for this study.

Future improvements

Future work could incorporate a routed sediment budget or transport model (e.g., SedNet) to estimate downstream attenuation and deposition, enabling sediment loads to be spatially attributed to exposed beneficiaries and used to construct more realistic population- or ecosystem-weighted demand indicators for water purification.

An alternative approach would be to explicitly represent spatial variation in the marginal cost of sediment abatement across the landscape. This could support more efficient targeting of interventions by identifying locations where sediment reductions can be achieved at lower cost. However, spatial attribution of ecosystem service benefits to downstream beneficiaries is likely to provide greater policy value because it directly links ecosystem service supply with the people and assets that receive the resulting benefits.

With additional time and resources, a CLUES-based approach could potentially be used to represent microbial attenuation and provide a spatial indicator of water purification related to E. coli.

5.6. Nutrient regulation

Explanation

Nutrient regulation uses outputs of the InVEST NDR model for supply and flow indicators.

The supply indicator for nutrient regulation is defined as the proportion of nitrogen and phosphorus retained by the landscape relative to the applied load, representing the intrinsic capacity of ecosystems to regulate nutrients independent of downstream water quality pressure. This efficiency-based measure captures differences in land cover, soils, and hydrological connectivity and is therefore well suited for comparing nutrient regulation capacity across space and scenarios.

The flow of the nutrient regulation service is represented by retained nutrients because they quantify the amount of nitrogen and phosphorus actually prevented from reaching waterways after accounting for landscape connectivity and retention processes. To derive a single indicator from both nutrients, the National Objectives Framework (NOF) band A guideline concentrations are used as reference values to weight retained nitrogen and phosphorus:

$$Flow = 0.5\left(\frac{TN_{ret}}{C_{TN,ref}}\right) + 0.5\left(\frac{TP_{ret}}{C_{TP,ref}}\right)$$

Where $C_{TN,ref}$ = 0.16 (Total nitrogen (trophic state)) and $C_{TP,ref}$ (Total phosphorus (trophic state). The lake trophic state guidelines represent low-impact, protective concentrations at the receiving-environment scale. The river-specific guidelines are for nitrates and dissolved reactive phosphorus, which are not specifically modelled by Invest NDR.

Demand: downstream water bodies impacted by nutrient loads – Rivermaps modelled water quality.

Total Nitrogen (TN): ≤ 1.0 mg/L

Total Phosphorus (TP): ≤ 0.02 mg/L

Monetisation

The nutrient regulation service is valued using an avoided-cost approach, which estimates the cost that would be incurred to achieve equivalent reductions in nutrient losses using engineered mitigation measures. In principle, the value of nutrient retention would ideally be linked to the improvement in downstream water quality, for example by modelling how reductions in nitrogen and phosphorus loads translate into changes in in-stream concentrations and ecological outcomes. However, this requires a hydrological connectivity model capable of tracing nutrient transport from each location through the river network to receiving environments, which is beyond the scope of the current analysis. The UN ecosystem accounting guidelines and much of the ecosystem services literature explicitly recognise avoided abatement cost as a pragmatic proxy when linking ecosystem functions to final environmental outcomes is not feasible (United Nations, 2022).

Matthews, Craggs, Tanner and Holland (2024; 2026) estimated marginal abatement costs for nutrient reductions using optimised bundles of edge-of-field mitigation systems (e.g., riparian planting, detainment bunds and constructed wetlands) applied across a range of rural landscapes. The estimated costs vary with soil permeability and the level of removal required, but for a moderate 20% reduction in nutrient losses the marginal abatement costs average $44 per kg N and $162 per kg P.

These cost ranges are consistent with an earlier study by Daigneault et al. (2016) that simulated policies that place a price on nitrogen leaching. They reported marginal abatement costs, based on loss of farm profit or additional management costs, in the range of NZ$10–30 / kg N and $50-200 /kg P ($15-59 per kg N and $73-293 per kg P in current prices).

A complication is that mitigation systems targeting one contaminant often also reduce the other. As a result, the marginal costs for nitrogen and phosphorus cannot simply be added when valuing simultaneous retention. Tanner and Matthews (2026) estimate the proportion of the other nutrient that is removed when mitigation systems are optimised to reduce a particular contaminant. The table below, derived from the forthcoming paper's supplementary data, shows the marginal abatement cost for different reduction targets and the associated co-benefit reductions in the other nutrient, averaged across landscape types.

Table 4: Marginal abatement cost curves and co-benefits for nutrient mitigation, derived from Matthews et al. (2026)

Target reduction	P co-benefit	N co-benefit	Mid $/kg N	Low $/kg N	High $/kg N	Mid $/kg P	Low $/kg P	High $/kg P
10%	15%	3%	28	21	36	142	107	176
20%	32%	6%	44	36	52	162	130	195
30%	34%	10%	43	36	51	169	122	217
40%	41%	12%	46	38	54	174	119	228
50%	54%	10%	50	42	58	152	90	213
60%	70%	11%	55	47	64	151	87	214
70%	93%	12%	60	51	69	159	92	226

A full analysis would involve running the mitigation optimisation model for every catchment to account for the differences in permeability. However, this analysis implements a shortcut using the table above to evaluate just two abatement strategies: (i) systems optimised for nitrogen reduction and (ii) systems optimised for phosphorus reduction. The co-benefit reductions are used to determine how much of the second nutrient would already be removed under each strategy, and the remaining reduction required is valued using the corresponding marginal cost curve. The total cost of each strategy is calculated, and the minimum of the two is used as the estimated abatement cost.

Data sources

- LRIS Water Yield
- Iris-nitrate-leaching
- Iris-new-zealand-land-use-management-version-03-nzlum-v03

InVEST NDR model inputs

Nutrient run-off proxy

LRIS Water yield is the best available nutrient runoff proxy because it integrates precipitation, evapotranspiration, soils, and vegetation into a spatially explicit measure of effective runoff generation, rather than representing rainfall alone.

Land-use map

Land Use Management (NZLUM v03) layer is better suited to InVEST NDR than LCDB6 because it distinguishes managed land uses that actually drive nitrogen loads (for example dairy, sheep and beef, arable, horticulture). LCDB6 tends to lump very different nutrient sources into broad classes.

The unique ID for land use classes is based on lu_code and first commodity (which distinguishes dairy from other grazing. Only the first commodity is used, otherwise it would make too many unique classes.

Steps:

- Add a calculated field in NZLUM for the first commodity (multiple commodities makes too many classes).
- Add a calculated field to create a unique lu_code and first commodity class
- Create a csv with numeric codes for each of these unique classes.
- Add a calculated field lu_class_num by joining on this csv

- Rasterize the class number

Biophysical table

The LRIS nitrate leaching layer is a strong basis for calculating land-use-specific load_n values because it provides spatially explicit, nationally consistent estimates of nitrogen leached below the root zone.

There is no comparable layer available for phosphorus losses. Therefore, indicative values for each land use category are derived from a variety of published literature (Matheson et al., 2024; Monaghan et al., 2021; Thomas et al., 2011).

To calculate load_n:

- Use zonal statistics to get mean NLeach for every feature in NZLUM
- Average NLeach for each lu_class_num

Other biophysical table parameter values were selected to preserve relative differences between land uses, remain internally consistent, and reflect New Zealand process understanding, while avoiding false precision where empirical, spatially explicit data are unavailable.

Retention efficiencies (eff_n, eff_p):

- Represent the maximum fraction of nutrients retained by a land-use class once the critical length is reached.
- Values (0-1) were assigned by land-use intensity:
    - High retention for indigenous vegetation, forests, and conservation land.
    - Moderate retention for pasture and perennial horticulture.
    - Low retention for intensive cropping, urban, mining, and highly disturbed land.
- Nitrogen efficiencies were generally lower than phosphorus efficiencies, reflecting weaker and more variable N retention in soils relative to P sorption.

Critical lengths (crit_len_n, crit_len_p):

- Define the downslope distance required for a land-use type to reach its maximum retention capacity.
- Values were chosen as multiples of the 100 m raster resolution to avoid sub-pixel interpretation.
- Assigned by land-use class:
    - Long critical lengths for forests and natural vegetation, reflecting cumulative retention along vegetated flow paths.
    - Intermediate lengths for pasture and cropping systems.
    - Short lengths for urban, mining, and bare land, where maximum (low) retention is reached quickly.
- Phosphorus critical lengths were generally shorter than nitrogen, reflecting more rapid sorption and particulate trapping processes.

Proportion of subsurface nitrogen transport (proportion_subsurface_n):

- Specifies the fraction of nitrogen load transported via subsurface pathways, with the remainder assumed to travel via surface runoff.
- Assigned based on dominant nitrogen loss pathways:
    - High subsurface fractions for pastoral land, forestry, and horticulture, where nitrate leaching dominates.
    - Mixed fractions for cropping and intensive animal systems.
    - Low subsurface fractions for urban, mining, and disturbed land, where runoff dominates.
- This approach reflects New Zealand evidence that riverine nitrogen loads are predominantly delivered via shallow groundwater rather than event runoff.

Fixed parameters

- Subsurface critical length (nitrogen): 200 (global average).
    - New Zealand evidence shows nitrogen is primarily transported via shallow subsurface flow (Yang et al., 2025) with substantial attenuation over short hillslope distances, and no NZ-specific empirical estimates exist to justify deviation from the global average.
- Subsurface Maximum Retention Efficiency: 0.8 (global average)
- Flow direction algorithm: Multiple Flow Direction MFD
    - MFD is generally preferred for InVEST NDR because it better represents diffuse hillslope flow and nutrient spreading typical of New Zealand landscapes, whereas D8 tends to over-concentrate flow into single paths and exaggerate delivery along artificial channels
- Threshold Flow Accumulation: 100 (≈ 1 km² at 100 m resolution).
    - This is a standard, defensible choice that approximates the scale at which channels become defined in New Zealand catchments and avoids over-densifying the stream network, while remaining consistent with typical InVEST NDR applications
- Borselli K Parameter: 2 (default)
- Average Runoff Proxy: blank (automatically calculated from the runoff raster)

Steps

- Use the InVEST NDR outputs load_n, load_p, surface_export_p, total_export_n
- Calculate retained_n = max(0,load_n - total_export_n)
- Calculate retained_p = max(0,load_p - surface_export_p)
- Run the Python script *Nutrient_regulation_calculate_cost.py*

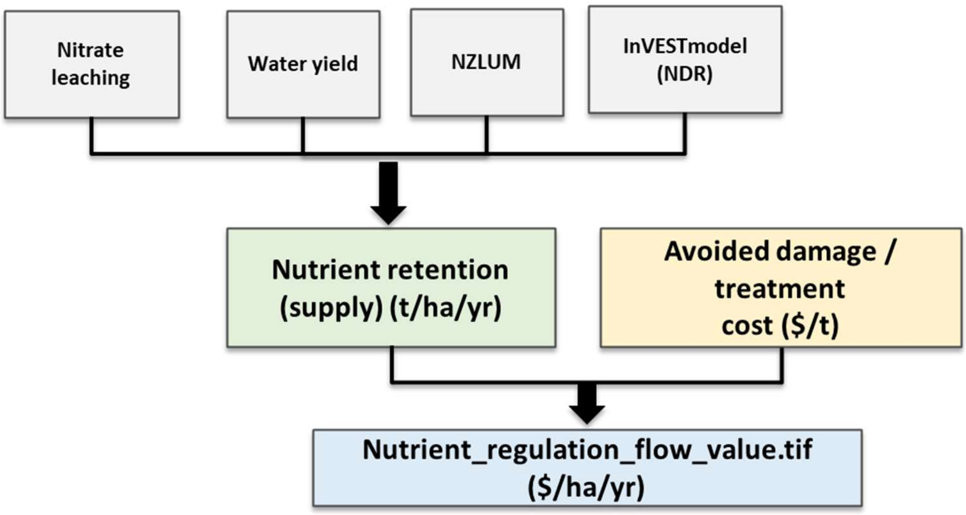

Figure 4: Flow diagram of nutrient regulation

Limitations

Land use categories could be partitioned by region to reflect regional differences in average N&P loads. However, there is a lot of intra-regional variation as well so this approach was not adopted, and nationally representative land-use averages were used instead to avoid introducing false precision.

Future improvements

A potential future improvement would be to apply catchment-specific nitrogen and phosphorus weights based on limiting nutrient status or receiving-water sensitivity, allowing the flow indicator to better reflect local ecological responses rather than assuming uniform nutrient importance across all catchments.

5.7. Erosion regulation

Explanation

Avoided erosion is estimated using the New Zealand Empirical Erosion Model (NZEEM) (Dymond et al., 2010) as a baseline, combined with land cover and vegetation information to distinguish between areas where vegetation effects are already represented and where additional mitigation can occur.

NZEEM provides estimates of current erosion rates under existing land cover conditions. In the model, woody vegetation substantially reduces erosion relative to non-woody land cover through a lower cover factor. As a result, erosion rates in areas classified as woody are interpreted as already reflecting the mitigating effect of vegetation.

A binary woody mask was derived from LCDB6 land cover classes. Pixels classified as woody vegetation (e.g. indigenous forest, exotic forest, scrub) are treated as "post-treatment" in NZEEM, meaning that the lower cover factor (1) has already been applied. For these areas, avoided erosion is inferred as the difference between the current (woody) erosion rate and a counterfactual non-woody state (cover factor 10). This reflects the erosion that would occur if woody cover were removed.

For non-woody LCDB classes (e.g. pasture, cropland), NZEEM erosion rates are treated as representing a "pre-treatment" state. In these areas, avoided erosion is estimated using two additional inputs:

- a mitigability factor derived from NZLRI erosion types, representing the effectiveness of vegetation in reducing different erosion processes; and
- a tree cover layer (0–100%), representing the proportion of woody vegetation present within the pixel.

The mitigability layer was developed from the New Zealand Land Resource Inventory (NZLRI) erosion attributes, which describe both the dominant erosion processes and their severity within each mapping unit. Each NZLRI polygon can contain up to four erosion types, each with an associated severity class. These erosion types were extracted and assigned vegetation effectiveness factors based on the review by Phillips et al (2020). For each polygon, the erosion types were combined into a single mitigability value by weighting the effectiveness factors according to their relative severity, with more severe or dominant erosion processes contributing more strongly to the final score. The resulting mitigability index ranges from 0 to 75% and represents the proportion of erosion that could be reduced through vegetation, independent of whether vegetation is currently present.

Erosion form	Space-planted tree effectiveness	Central estimate
Debris avalanche	Low–Moderate (20–50%)	35%
Earthflow	Low (0–20%)	10%
Earth slip	High (50–80%)	65%
Mudflow	Low–Moderate (10–40%)	25%
Soil slip	High (60–90%)	75%
Gully	Moderate (30–70%)	50%
Rill	Low (0–30%)	15%

Rockfall	Very low (0–10%)	5%
Riparian slip	Moderate–High (40–80%)	60%
Streambank	Moderate–High (40–80%)	60%
Slump	Low (0–30%)	15%
Tunnel gully	Low–Moderate (10–40%)	25%
Scree	Very low (0–10%)	5%
Sheet	Low (0–30%)	15%
Wind	Moderate (30–70%)	50%

The mitigability factor is scaled by a tree cover adjustment factor that increases from 0 when tree cover is absent to 1 when tree cover reaches 30%. This allows partial mitigation effects to be represented in landscapes where trees are present but not dominant, such as areas with pole planting, riparian vegetation, or scattered hillside trees.

This hybrid approach ensures that mitigation is not double-counted in areas already classified as woody in NZEEM, while capturing the additional contribution of sparse or partial tree cover in predominantly non-woody landscapes that is not explicitly represented in NZEEM.

Monetisation

The monetisation of erosion regulation was based on the approach developed by Soliman and Walsh (Soliman & Walsh, 2005), who estimated the marginal economic cost of soil erosion by linking erosion-induced productivity losses to farm profitability. In their framework, soil erosion reduces agricultural productivity, which in turn reduces farm profit; the marginal cost of erosion per tonne was therefore derived by dividing the estimated profit loss per hectare by the erosion rate. Because the resulting values are proportional to farm profit per hectare (EBIT), this study assumed the same relationship between marginal erosion cost and EBIT as in Soliman and Walsh (2005), but updated the underlying profit estimates to reflect contemporary conditions. The cost per tonne of erosion ranges from around $0.12 for low-intensity sheep and beef to $5 for fruit orchards.

Table 5: Original Soliman & Walsh EBIT and marginal costs, and updated EBIT and marginal cost.

Farm type	2005 EBIT/ha	2026 EBIT/ha	Marginal erosion cost 2005 ($/t)	Marginal erosion cost 2026 ($/t)
Arable	1650	2000	0.69	0.84
Dairy	3208	2845	1.64	1.45
Deer	586	800	0.29	0.40
Exotic forestry	612	900	0.13	0.19
Fruit	7406	12000	3.11	5.04
Native	0	0	0.00	0.00
Other	0	0	0.00	0.00
Other pasture	17.1	120	0.01	0.07
Sheep & beef	7.63	300	0.00	0.12
Vegetables	9356	10000	3.93	4.20

As the study did not report confidence intervals or other measures of uncertainty, lower and upper bounds were approximated using the adjacent farm-type estimates in the reported cost distribution, with the next lowest-cost farm type used as the lower bound and the next highest-cost farm type used as the upper bound.

Land-use classes were defined using the New Zealand Land Use Map (NZLUM), which was used to assign the appropriate marginal erosion cost values spatially. Dairy and sheep-and-beef profits were obtained from the Land Use Opportunities profit layers, while profits for other land uses were compiled from recent industry reports and sector benchmarks. Urban areas and non-productive land classes were excluded from the analysis so that erosion values were only applied to productive land where soil loss affects economic output.

Data sources

- lris-nzeem-erosion-rates (https://lris.scinfo.org.nz/layer/48178-nzeem-erosion-rates-north-island/, https://lris.scinfo.org.nz/layer/48176-nzeem-erosion-rates-south-island/)
- lris-nzlri-erosion-type-and-severity(https://lris.scinfo.org.nz/layer/48054-nzlri-erosion-type-and-severity/)
- Landcover database v.6 (LCDB6)
- Landuseopportunities dairy-operating-profit-v1
- Landuseopportunities sheep-and-beef-operating-profit-v1
- lris-new-zealand-land-use-management-version-03-nzlum-v03

Steps

- Combine NZEEM North and South islands, convert to t/ha, and warp to project grid.
- Create a binary raster representing woody areas using LCDB classes 52,54,56,58,64,68,69.
- Perform the following raster calculation and save as avoided_erosion.tif

```
where(@nzeem>0, where(@woody==1, 9*@nzeem, @nzeem *@mitigability*where(@treecover<15, 0, where(@treecover <30, (@treecover -15)/15.0, 1)) ),0)
```

- Run the Python script *Erosion_regulation_assign_cost.py* to assign mitigation cost based on NZLUM class

5.8. Fluvial and pluvial flood regulation

Explanation

Supply indicator

The flood regulation supply indicator represents the biophysical capacity of the landscape to attenuate, store, and slow storm runoff before it contributes to damaging peak flows. It is constructed as the multiplicative product

$$FLOOD\_SUPPLY = PERM \times ROUGH \times TWI \times SLOPE \times DIST$$

where each component captures a necessary element of flood attenuation capacity and is normalised to the 0-1 range. PERM represents soil infiltration capacity derived from the LRIS permeability profile, transformed to a unitless score reflecting the ability of soils to absorb and transmit water during high-intensity rainfall.

ROUGH represents surface roughness and interception derived from LCDB land cover classes, with forests, shrublands, and wetlands assigned high values and impervious urban surfaces assigned values near zero.

TWI represents topographic convergence, calculated from a topographic wetness index and normalised to emphasise areas where runoff naturally accumulates and interception is most effective.

SLOPE is a slope attenuation index derived from the LENZ slope raster using a nonlinear exponential transformation, with flat areas retaining high values and steep terrain strongly penalised to reflect rapid runoff response.

DIST is a normalised distance-to-stream index, where areas closer to the drainage network have higher values, reflecting greater opportunity to intercept runoff before it concentrates in channels. The multiplicative structure ensures that flood regulation supply is high only where all controlling factors align, and that any limiting factor, such as steep slopes or impervious cover, appropriately constrains overall capacity.

Demand indicator

Flood damage potential was represented using a probability-damage weighting derived from an expected annual damage framework, consistent with NIWA and GNS Science flood risk modelling practice (e.g. RiskScape), where damage severity increases nonlinearly with event rarity and losses are integrated over the flood frequency distribution.

A national flood map is not yet available[2], so the layer lris-fsl-flood-return-interval and FLOOD_CLASS (Webb & Willson, 1995) is used as an indicator of relative flood risk. The flood exceedance threshold *T* associated with each flood class is converted to a cumulative risk density for events exceeding *T* using an empirical damage potential function derived from the Insurance Council database of event costs (Denne & Wright, 2017).

Table 6: Risk density for FSL flood class

FLOOD_CLASS	Description	Flood return interval (years)	Cumulative risk density for T > x
1	Nil	Nil	0
2	Slight	< 1 in 60	0.12444
3	Moderate	1 in 20–1 in 60	0.24771
4	Moderately severe	1 in 10–1 in 20	0.38145
5	Severe	1 in 5–1 in 10	0.56505
6	Very severe	> 1 in 5	0.84315

Exposure is separated into property-related damage and land-based production damage. Property damage is also used as a proxy for exposure of associated infrastructure, including roads, water supply, electricity, and telecommunications, on the basis that these assets are typically spatially co-located with concentrations of property-based capital. An uplift factor of 20% is applied to account for infrastructure, since infrastructure typically adds 10-40% to direct damage in urban areas (European Commission. Joint Research Centre., 2017)

For land-based production damage, maximum annual damage per hectare was assumed to equal the loss of one year's net agricultural production plus immediate remediation costs, with values differentiated by LCDB class:

- Cropland (30): $8,000 per ha per year. One season's net crop margin lost plus re-sowing and soil remediation.
- Orchard & Vineyard (33): $ 20,000 per ha per year. Loss of annual harvest and clean-up in high-value perennial systems, excluding tree or vine replacement.
- High Producing Grassland (40): NZD 2,000 per ha per year. Lost grazing and pasture damage in intensive pastoral systems.
- Low Producing Grassland (41): NZD 900 per ha per year. Short-term feed loss and minor pasture repair in extensive systems.

Potential flood damage to forest land was not included. While extreme events can cause localised tree loss, commercial forestry in New Zealand is predominantly located on hill country outside floodplains, and flood-related impacts rarely translate into measurable annual economic losses at the scale considered here.

Flow indicator

In the absence of hydrological or hydraulic modelling, and without any existing basis for retention into volumes or peak flow reductions, realised flood regulation is operationalised

[2] https://environment.govt.nz/what-government-is-doing/areas-of-work/climate-change/adapting-to-climate-change/national-adaptation-framework/national-flood-map/

using a simplified flow indicator. Sea-draining catchments define the maximum spatial extent over which upstream flood regulation benefits can accumulate.

The indicator is calculated for each sea-draining catchment as the product of catchment-average regulatory capacity, demand (residual flood risk-weighted asset exposure), and a conservative effectiveness cap. Average upstream supply represents the relative capacity of landscapes to attenuate runoff through infiltration, roughness, storage, and hydrological connectivity, while demand captures the magnitude of remaining flood risk to exposed assets.

The literature on nature-based flood management report peak flow or damage reductions typically below 30 percent at catchment and regional scales, with many studies reporting effects below 10 percent (OECD, 2020). For this rapid national assessment, the contribution of ecosystem-based flood regulation is conservatively capped at 20 percent of residual risk to avoid overstating protection.

The calculation applied for flow in cell *i* and catchment *c* is:

$$Flow_i = 0.2\ Demand_i\ \overline{\mathrm{Supply}_{c(\iota)}}$$

The resulting raster therefore represents the expected value of avoided flood damages attributable to ecosystem-mediated regulation, scaled spatially by both exposure and regulatory capacity.

Limitations

The Iris-fsl-flood-return-interval layer largely excludes urban areas, which means capital exposure is likely to be underestimated. Conversely, the layer does not explicitly represent flood protection schemes, potentially overstating exposure in defended locations. The results should be interpreted as an indicative spatial signal of where flood attenuation benefits are most likely to be realised, rather than as an estimate of absolute flood risk or damage.

Changes in water retention influence downstream flood risk primarily through attenuation of peak flows rather than reductions in total runoff. While conversion of retention into changes in flood frequency would require event-based hydrological and hydraulic modelling, such analysis is beyond the scope of this study. Instead, retention is interpreted as reducing the probability of exceedance of damaging flood thresholds downstream, consistent with NZ flood risk modelling practice.

Asset exposure and potential damage is represented at a more aggregated level than in platforms such as RiskScape[3], which utilise detailed, asset-specific layers. The approach adopted here is intended to provide an indicative, spatially consistent assessment rather than a detailed loss estimation.

Future improvements

- Incorporate the national flood map when it becomes available
- Conduct a national Riskscape analysis to calculate damage across a range of return intervals, enabling more precise estimation of avoided and residual risk.

Future work could investigate the extent to which changes in catchment-scale water retention translate into reductions in downstream flood exceedance probabilities. This would require event-based hydrological modelling using established New Zealand tools such as TopNet-NZ or HydroTel, coupled with hydraulic routing models to assess peak flow attenuation at downstream assets. Repeating such analyses across a range of design events or long rainfall sequences would enable changes in flood frequency to be quantified explicitly.

[3] https://www.riskscape.org.nz/

The damage to forestry estates from flooding can be seen more in infrastructure losses and repairs. The costs associated with roading, culvert, bridge and fencing wash-outs can be substantial and in more vulnerable districts on-going maintenance will be required to maintain the infrastructure to a level required for harvesting operations. This could be another area for future attention when workable data becomes available.

Data sources

- Iris-fsl-permeability-profile
- Iris-nzenvds-topographic-wetness-index-v10
- Iris-lenz-slope
- Iris-lcdb-v60-land-cover-database-version-60-mainland-new-zealand
- statsnzstatistical-area-1-2023-generalised

Steps

Create supply indicator

- Convert permeability to a score ranging from 0 (town) to 1 (rapid)
- Normalise topographic wetness index (TWI) to a range 0 to 1
- Convert slope degrees to a unitless attenuation index using a capped exponential decay function, assigning maximum flood attenuation capacity to very flat terrain (≤2°), strongly penalising steep slopes (≥45°):

```
if( "lenz-slope@1" <= 2, 1, if( "lenz-slope@1" >= 45, 0.05, exp(-0.06 * "lenz-slope@1") ))
```

- Convert LCDB class to a unitless roughness score by assigning higher values to vegetation types with greater interception and flow resistance, and rasterise.

Table 7: Mapping of LCDB6 2023 land-cover classes to unitless surface roughness scores (0–1)

Class_2023	Description	Roughness score
1, 5	Built-up; Transport infrastructure	0
2	Urban park	0.3
6	Mines & dumps	0.1
10, 16	Sand & gravel; Gravel & rock	0.15
12	Landslide	0.2
14	Snow & ice	0.05
30	Cropland	0.45
33	Orchard & vineyard	0.5
40, 41	High- and low-producing grassland	0.6
43, 44, 15	Tussock grassland; Depleted grassland; Alpine grass	0.55
45, 46	Herbaceous freshwater; Herbaceous saline	0.9
47	Flaxland	0.85
50	Fernland	0.8
70	Mangrove	0.9
51	Gorse & broom	0.7
52, 54, 55, 56, 58	Mānuka & kānuka; Broadleaved indigenous; Sub-alpine shrubland; Mixed exotic shrubland; Grey scrub	0.8
69, 71, 68	Indigenous forest; Exotic forest; Deciduous hardwood	0.85
64	Forest harvested	0.65
20, 21, 22	Lake & pond; River; Estuarine	NULL

- Multiply rasters PERM×ROUGH×TWI×SLOPE×DIST

- Because products of many small terms quickly collapse to 0, convert to natural log and then normalise to preserve relative differences.

Create demand layer

- Add to properties points a calculated attribute for improved value (capital minus land)
- Cap improved value at the 99.9th percentile ($10.6 million in this case) to remove data anomalies.
- Add a calculated field exposed_property =

```
min(10640000,improved_value) *
CASE
 WHEN "FLOOD_CLAS" = 1 THEN 0
 WHEN "FLOOD_CLAS" = 2 THEN 0.124444998
 WHEN "FLOOD_CLAS" = 3 THEN 0.247707723
 WHEN "FLOOD_CLAS" = 4 THEN 0.381449043
 WHEN "FLOOD_CLAS" = 5 THEN 0.565046331
 WHEN "FLOOD_CLAS" = 6 THEN 0.843149468
 ELSE 0
END
```

- Use “join to features by location (summary)” to sum exposed property by SA1
- Add a calculated attribute for exposed property per hectare of land, multiplying by the 20% uplift factor.
- Rasterise and save as exposed_capital.tif
- Calculate land-based production damage from the LCDB raster as: A*((B==30)*8000 + (B==33)*20000 + (B==40)*2000 + (B==41)*900) where A = residual flood risk and B = Class_2023. Save as exposed_land.tif
- Add both exposed_capital and exposed_land and save as flood_regulation_supply.tif

Create flow indicator

- Sum demand value by catchment
- Run the python script *Flood_regulation_allocate_value.py* to allocate value to supply cells, weighting by supply.

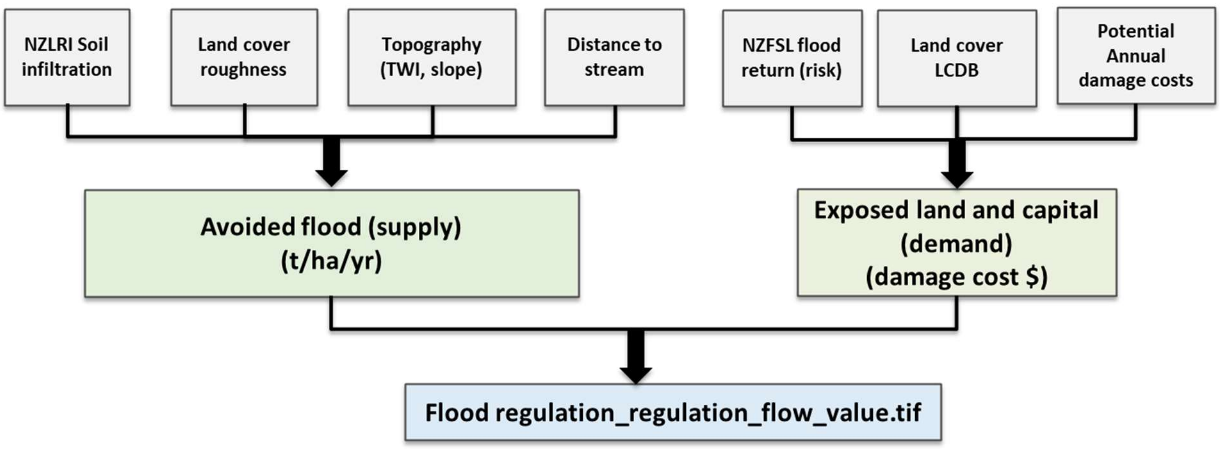

Figure 5: Flow diagram for flood regulation

5.9. Coastal storm surge regulation

Explanation

This layer represents the ecosystem service provided by mangroves and saltwater wetlands in reducing coastal hazard impacts, primarily through attenuation of wave energy and storm surge. These ecosystems act as natural buffers that dissipate wave energy, increase surface roughness, and reduce water levels during storm events. This results in lower flood depths, reduced flood extents, and decreased shoreline erosion, ultimately reducing damages to exposed assets.

Empirical and modelling studies show that coastal wetlands can significantly reduce flood damages and hazard exposure. For example, Narayan et al. 2017 estimate that coastal wetlands reduce global flood damages by over US$100 billion annually, while Barbier et al. 2013 demonstrate that wave attenuation and surge reduction increase with wetland extent but exhibit diminishing returns with distance. Site-specific hydrodynamic modelling by Fairchild et al. (2021) shows that the presence of saltmarsh can reduce 1-in-100-year flood damages by up to approximately 37%, providing an upper-bound estimate of effectiveness in large, continuous systems.

In this dataset, the coastal hazard regulation service is quantified for the flood damage component using a spatially explicit avoided damage approach. The method proceeds as follows:

Hazard exposure and demand

A 1-in-100-year coastal flood extent is used to define exposed areas. Property capital values within the flood extent are rasterised and used to estimate expected annual flood damage. This serves as a proxy for broader economic damages, including impacts to public infrastructure and non-market or intangible losses, which are not explicitly modelled.

Flood extent aggregation

Flood extents are represented as polygons. For each flood extent feature, total expected annual damage is calculated and then averaged across all raster cells within that feature to produce a spatially uniform demand surface.

Ecosystem service supply

Mangrove and saltwater wetland extent are represented as binary rasters. Within each flood extent feature, the proportion of wetland area is used as a proxy for the capacity of ecosystems to attenuate coastal hazards.

Effectiveness and avoided damage

Flood damage reduction is estimated as a function of wetland proportion:

$$r(P) = \min(\beta P,\, r_{\max})$$

where $P$is the wetland proportion within the flood extent, $\beta$is an effectiveness parameter, and $r_{\max}$is a cap on maximum reduction. The cap is informed by literature (e.g. Fairchild et al. 2021). A distance-decay adjustment is applied to reflect diminishing protection with increasing distance from wetlands.

Allocation of benefits (realised flow)

The total avoided damage within each flood extent is calculated and then allocated evenly across all mangrove and wetland cells within that feature. This produces a realised flow layer representing the spatial distribution of ecosystem service benefits attributable to coastal ecosystems.

Three scenarios are used to reflect uncertainty in both flood damages and wetland effectiveness, representing low, central, and high estimates. The damage ratio (5–20%) represents the proportion of property capital value lost in a 1-in-100-year event, spanning lower-bound shallow flooding through to more severe impacts. The effectiveness parameter (β) controls how strongly wetland extent translates into damage reduction, with lower values representing fragmented or narrow wetlands and higher values representing more continuous systems. The maximum reduction places an upper bound on total damage reduction, ensuring results remain consistent with empirical evidence that wetlands reduce, but do not eliminate, coastal flood damages. The low scenario is deliberately conservative and reflects limited wetland influence, while the central and high scenarios are informed by empirical and modelling studies showing that wetlands can materially reduce flood damages where they are sufficiently extensive (Barbier et al., 2013; Fairchild et al., 2021; Narayan et al., 2017).

Table 8: Parameters for low, central and high scenarios of coastal hazard regulation

Scenario	Damage ratio	Effectiveness β	Maximum reduction	Support
Low	5%	0.25	5%	Consistent with partial or thin wetlands; conservative vs global studies
Central	10%	0.5	10%	Supported by Narayan et al. and Barbier et al. for meaningful wetland presence
High	20%	1	20%	Directly from Fairchild-type modelling studies (site-specific upper range)

Data

- Seasketch habitats (https://seasketch.doc.govt.nz/arcgis/rest/services/National/Our_Estuaries/MapServer/76)
- National sea-level rise 1-in-100 year flood extent (SLR-0) (https://data.niwa.co.nz/products/national-sea-level-change)
- Council property data (2020 snapshot)
- lris-lcdb-v60-land-cover-database-version-60-mainland-new-zealand

Steps

- Rasterise LCDB6 class “Herbaceous Saline” and “mangrove”
- Run the Python script *natural_hazard_coastal_flooding_calculate_flow.py*

Limitations

This layer represents a screening-level estimate of the flood damage reduction component of coastal hazard regulation. It does not explicitly model hydrodynamic processes such as flow routing, water levels, or sediment transport, and therefore does not capture all aspects of coastal protection, particularly long-term shoreline erosion dynamics. Results are sensitive to assumptions regarding damage ratios and ecosystem effectiveness, and should be interpreted as indicative of the order of magnitude and spatial distribution of benefits rather than precise estimates.

The analysis is based on a single 1-in-100-year coastal flood extent, which is the only nationally consistent dataset available for this application. In practice, a more robust assessment would consider a range of flood events with different probabilities (e.g. 1-in-10, 1-in-50, 1-in-100, and extreme events) to better capture the full distribution of expected damages and the contribution of ecosystems across varying hazard intensities.

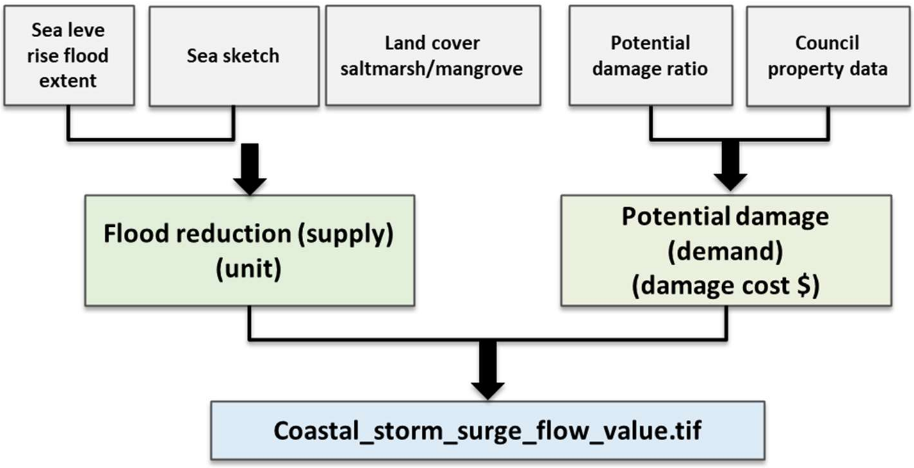

Figure 6: Flow diagram for coastal storm surge regulation

5.10. Pollination

Explanation

The pollination indicator focuses on wild insect pollination because it represents an ecosystem service delivered by unmanaged biological processes, whereas managed pollination is a human input that supplements but does not constitute a natural ecosystem service. Pollination services were mapped using the InVEST Crop Pollination model. Managed pollination was parameterised only to account for its interaction with wild pollinators, but the analysis and valuation explicitly isolate the contribution of wild insects to crop production.

Land use and crop representation

Land use and land-based production were classified using the commodities attribute of the New Zealand Land Use Map (NZLUM). Only crops with a non-zero dependence on insect pollination were included in the farm layer used by the model. Land uses dominated by grazing were excluded from pollination demand, as pasture production in New Zealand is assumed to be primarily ryegrass-based and therefore wind-pollinated, with no direct dependence on insect pollinators. Some pasture is mixed clover, but there is no spatial data showing where clover pasture is. Pasture land uses nevertheless contribute indirectly to pollination supply through habitat and narrow linear features such as shelter belts, paddock margins, road verges, and other flower-rich areas, which are captured in the biophysical inputs.

Representation of pollinators and habitat

Wild pollination services were represented using two functional groups of insect pollinators: ground-nesting and cavity-nesting insects. This abstraction captures the dominant nesting strategies of wild pollinators in New Zealand, including native bees, flies, and other generalist insects.

Pollinator populations are supported by two key habitat components: nesting suitability and floral resource availability. Nesting suitability reflects the availability of appropriate substrates for each nesting type, while floral resources represent the availability of nectar and pollen during the relevant flowering seasons.

A biophysical table was constructed to link NZLUM land-use classes to pollinator habitat suitability. Each land-use class was assigned relative indices for ground-nesting suitability, cavity-nesting suitability, and seasonal floral resource availability (spring and summer). Values were assigned on a relative scale from 0 to 1, reflecting contrasts between land uses rather than absolute ecological quantities. Assignments were informed by land-cover characteristics, vegetation structure, and expected disturbance intensity, following the established MAES pollination modelling approaches (Stange et al., 2017; Zulian et al., 2013)

Crop attributes and pollination response

Crop-specific attributes were defined in a crops table, including the half-saturation coefficient (half_sat), pollinator dependence (p_dep), and the proportion of pollination supplied by managed pollinators (p_managed). These parameters determine how crop yield responds to pollinator abundance.

Pollinator dependence values were based on published global syntheses of crop pollination dependence, while half-saturation coefficients were set within recommended ranges for InVEST applications (Kennedy et al., 2013; Zulian et al., 2013).

Indicators of supply, demand, and realised flow

Pollination supply was represented by the maximum seasonal wild pollinator abundance across spring and summer. Using the maximum rather than a seasonal average reflects the landscape's capacity to support pollinators during the most favourable period, avoids underestimating seasonally important habitats, and aligns with the ecological reality that pollinator populations are constrained by peak resource availability rather than mean conditions.

Pollination demand was defined at the 1 ha cell level as the product of crop pollinator dependence (p_dep) times typical crop value. Indicative crop values were sourced from several industry publications, particularly Fresh Facts (United Fresh New Zealand, 2025). Because all cells have equal area and only dominant crop identity is known, this formulation represents relative demand per hectare under an assumption of uniform production intensity within crop types.

Realised ecosystem service flow was represented using the InVEST output y_wild, which measures the proportion of total crop yield attributable to wild pollinators. This indicator captures the actual contribution of wild pollinators to agricultural production under current landscape conditions and avoids double-counting pollinator-independent yield. It therefore provides an appropriate basis for ecosystem service valuation and spatial comparison.

Monetisation

The monetary value of wild pollination is estimated using a replacement cost approach, which reflects the cost that would be incurred if pollination services provided by wild insects had to be replaced with managed pollination. This method is commonly used in ecosystem service valuation when a market substitute exists for the service. In the case of crop pollination, growers frequently supplement natural pollination by hiring commercial honeybee colonies during flowering. The cost of introducing managed hives therefore provides a practical proxy for the economic value of the pollination service.

Average pollination rental fees were obtained from MPI's 2025 Apiculture Monitoring (https://www.mpi.govt.nz/resources-and-forms/economic-intelligence/farm-monitoring#apiculture) , which reports a mean cost of approximately NZ$200 per hive. To translate this into a per-hectare value, assumptions are required about typical hive stocking rates.

The number of hives required per hectare depends on the crop and conditions, but four is a typical stocking rate (Goodwin, 2012). Kiwifruit is an important exception because of its high pollen demand and relatively unattractive flowers. Kiwifruit orchards commonly deploy 8–12 hives per hectare to ensure adequate pollination (Broussard & Pattemore, 2023).

Based on these values, the analysis assumes four hives per hectare for most crops and ten hives per hectare for kiwifruit orchards. At a cost of NZ$200 per hive, this corresponds to an approximate pollination cost of NZ$800 per hectare for most pollinated crops and NZ$2,000 per hectare for kiwifruit.

Some crops grown under protective cover use bumblebees rather than honeybees for pollination. Commercial bumblebee colonies typically cost around NZ$150 per hive, and recommended stocking densities are approximately 6–9 hives per hectare, resulting in costs of $900-1350 per hectare. Because these costs are of a similar magnitude to the cost of honeybee pollination, the valuation does not distinguish between pollinator species and instead applies a common replacement cost framework across crop types.

The estimated pollination cost per hectare is multiplied by InVEST output y_wild to ensure that only the proportion of pollination services attributable to wild pollinators is monetised.

Limitations

The pollination model draws on parameters and relationships developed largely from European and North American studies. Although these provide a practical basis for national-scale modelling, their direct application to New Zealand introduces uncertainty because pollinator communities, crop associations, and landscape contexts differ substantially. In particular, New Zealand's native bee fauna differs markedly from those regions, and the contribution of many native species to crop pollination remains poorly understood. At the same time, populations of feral honey bees have declined in some areas due to Varroa and other pressures, making assumptions based on abundant feral honey bee populations less appropriate than they once were.

The demand layer is also subject to data limitations. The New Zealand Land Use Map (NZLUM) does not include all pollinator-dependent crops and has incomplete coverage of some high-value horticultural land uses, including berryfruit and cherries. As a result, demand for pollination services is likely to be underestimated in some locations. In addition, the layer focuses on commercial agricultural production and does not capture pollination demand associated with suburban and peri-urban gardens, orchards, or other small-scale food production.

Future improvements

Using Agribase would allow for the inclusion of smaller horticultural enterprises such as berries and cherries, greenhouses, and mixed enterprises.

Data

- lris-new-zealand-land-use-management-version-03-nzlum-v03
- lds-nz-coastlines-and-islands-polygons-topo-150k

Steps

- Create a numeric index for unique NZLUM lu_code and first commodity combinations. This needs to match the biophysical table.
- Rasterise the numeric commodity index and save as NZLUM.tif
- Select NZLUM lu_codes from 2.2.0 to 2.6.0 as a farms layer
- Use first commodity as the crop type but where it doesn't exist, set a default class based on level 2:

```
if( "commod_1" IS NULL,
 if("lu_code" = '2.3.0',
  'short_rotation_cropping_undefined',
  if("lu_code" = '2.4.0',
   'perennial_horticulture_undefined', 'grazing_undefined'
  ) ), "commod_1")
```

- Join the resulting commodity class on the farm attributes csv which has the fields half_sat, p_managed, season, fr_spring, fr_summer, n_cavity, n_ground, p_dep.
- Filter the polygon features to select only features where p_dep >0[4]
- Save as farms.shp
- Run InVEST Pollination model using NZLUM.tif as the landcover layer and farms.shp as the farms layer

Create the demand layer

- Using the farms.shp layer, multiply p_dep * crop value
- Rasterise and save as pollination_demand.tif

Create the supply layer

- Run the InVEST model again with no farms to get complete layers of total pollinator abundance
- Use raster calculator to calculate max of spring and summer abundance
- Mask to land outline
- Export as pollination_supply.tif

Create the flow layer

- Calculate avoided cost/ha as

```
if("croptype"="kiwifruit",2000,800)*"y_wild"
```

- Rasterise and export as pollination_flow_value.tif

5.11. Regulation of waste

Explanation

The regulation of waste ecosystem service reflects the ability of soils to break down and assimilate organic wastes, such as livestock dung, through biological decomposition processes. By transforming organic waste into stable soil organic matter and nutrients, this service reduces the accumulation of waste on land and limits associated environmental impacts.

The potential supply indicator represents the capacity of soils to biologically decompose organic waste, expressed as the proportion of the year that soil moisture conditions are suitable for aerobic microbial activity. Following the conceptual framework of Dominati et al. (2014), soil moisture suitability was represented as the proportion of the year that soils are neither water-limited nor waterlogged. In the absence of national-scale daily soil moisture data for recent years, this was approximated using annual water deficit as a proxy, scaled to a biologically plausible range of 0.3 to 0.75 with a small downward adjustment for poorly drained soils to account for waterlogging and oxygen limitation.

The indicator is calculated as:

```
max( 0.30, min( 0.75, 0.75 - 0.45 * max("nzenvds-annual-water-deficit-v10@1", 0) / max_deficit))
```

where max_deficit = 322mm/year in the NZEnvDS water deficit layer.

The demand indicator represents the pressure placed on soils to regulate organic waste, quantified as the amount of livestock dung deposited per hectare per year. It reflects the volume of waste requiring biological decomposition, independent of whether soils have

[4] This reduces the number of farm features from 2.4 million to around 200,000. The model had fatal memory problems with 2.4 million features.

sufficient capacity to process it. Dung deposition was estimated using faecal dry matter excretion rates derived from New Zealand pasture intake and digestibility assumptions (≈5 kg DM cow^{-1} d^{-1} for dairy; ≈0.4 kg DM RSU^{-1} d^{-1} for sheep and beef), consistent with Tier 2 livestock modelling frameworks. Stocking rates are estimated using the farm typologies described in Monaghan et al. (2021). Stocking rates could alternatively be calculated from Agribase if available.

The realised flow indicator represents the amount of organic waste regulated by soils, calculated as the quantity of dung deposited multiplied by the proportion of the year that soil moisture conditions are suitable for biological decomposition (p_moist). In this formulation there is no explicit upper limit on the quantity of dung. The indicator assumes that, whenever moisture conditions are suitable, decomposition proceeds without saturation or capacity constraints.

Monetisation

The realised flow of the waste assimilation service was valued using an avoided treatment cost approach. This method estimates the economic value of ecosystem processing of organic waste by reference to the cost of removing equivalent organic loads using engineered wastewater treatment systems. Soil microbial decomposition of dung reduces the organic load that would otherwise contribute to biochemical oxygen demand (BOD) in receiving waters. The value of the service was therefore approximated using the marginal cost of treating organic loading expressed as kilograms of BOD.

The equivalent organic loading (BOD_5) associated with cattle manure was sourced from the American Society of Agricultural Engineers (2005). These data indicate approximately 0.60 kg BOD_5 produced per dairy cow per day and about 60 kg fresh manure per cow per day. Assuming manure contains 12-15% dry matter, this corresponds to approximately 0.07 kg BOD_5 per kg manure dry matter.

The economic value of avoided organic waste treatment was derived from trade-waste charging schedules used by New Zealand wastewater utilities, which include charges for organic loading expressed in kilograms of BOD. These charges provide a proxy for the marginal cost of removing organic pollution in wastewater treatment plants. Reported values vary across councils, reflecting differences in plant size and treatment technology, with smaller systems typically facing higher marginal treatment costs. Examples include:

- Hamilton City Council: $0.80 per kg BOD
- Palmerston North City Council: $0.79 per kg BOD
- Manawatū District Council: $0.84 per kg BOD
- Taupō District Council: $0.80 per kg BOD
- South Waikato District Council: $1.21 per kg BOD
- Waipā District Council: $1.55 per kg BOD

Lower values are observed in some larger systems, such as Dunedin ($0.23 per kg BOD), where charges primarily reflect treatment costs rather than full network cost recovery. In Auckland, Watercare's trade-waste charging formula implies an organic loading coefficient of approximately $0.47 per kg BOD.

Based on these values, a central treatment cost of $0.80 per kg BOD was adopted for the valuation.

Combining the organic load coefficient and treatment cost gives a unit value of 0.80×0.07 = 0.056 per kg dung dry matter

Data

- lris-new-zealand-land-use-management-version-03-nzlum-v03
- nzenvds-annual-water-deficit-v10

- lris-nzlri-land-use-capability-2021

Steps

Supply raster:

- Convert annual water deficit to a time proportion:

```
max( 0.30, min( 0.75, 0.75 - 0.45 * max("nzenvds-annual-water-deficit-v10@1", 0) / 322))
```

- Adjust for drainage:

```
if("drain_class@1" = 1, "p_moist@1" * 1.2,
if("drain_class@1" = 2, "p_moist@1" * 1.1,
if("drain_class@1" = 3, "p_moist@1",
if("drain_class@1" = 4, "p_moist@1" * 0.9,
if("drain_class@1" = 5, "p_moist@1" * 0.8,
NULL)))))
```

- Multiply by 100 and export as Int16 data type waste_assimilation_supply.tif

Demand raster:

- Filter NZLUM for lu_description “Grazing modified pasture systems”
- Add a field for farm typology.
- Join on a csv with dung figures for each typology
- Rasterize and save as waste_assimilation_demand.tif.

Flow raster

- Calculate supply times demand:

```
"waste_assimilation_demand@1" * ("waste_assimilation_supply@1" / 100.0)
```

- Save as waste_assimilation_flow.tif
- Multiply by 0.056 and save as waste_assimilation_flow_value.tif

Limitations

This indicator is a simplified representation of soil waste assimilation. It uses annual water deficit and drainage class as proxies for decomposition conditions and does not account for seasonal variation, soil biology, temperature, soil properties, or management practices. Demand estimates are based on average stocking rates and excretion assumptions rather than actual livestock distributions. The realised flow calculation assumes decomposition increases linearly with suitable moisture conditions and does not account for capacity limits, nutrient accumulation, or other adverse effects that may occur under intensive stocking. The monetary valuation is also approximate, as wastewater treatment costs may not fully reflect the value of natural soil-based waste processing.

Future improvements

Potential improvements would include replacing the annual water deficit proxy with seasonal or daily soil moisture data, adding soil temperature and texture constraints, and using measured or modelled decomposition rates where available. The demand layer could also be improved using actual livestock distribution, grazing duration, and manure management data, rather than average farm typologies. A further improvement would be to add a capacity threshold so the model distinguishes between waste that is assimilated and excess organic or nutrient loading.

A potential improvement to the monetary valuation would be to link ecosystem service benefits more directly to downstream beneficiaries. Rather than valuing waste assimilation using generic wastewater treatment costs, future work could estimate the extent to which soil decomposition reduces organic loading reaching waterways and value the resulting benefits to downstream users.

5.12. Global climate regulation - blue carbon

The blue carbon regulating service refers to the role of coastal wetland ecosystems such as mangroves, saltmarsh, and seagrass in moderating atmospheric greenhouse gas concentrations through carbon storage and sequestration. These ecosystems capture carbon dioxide from the atmosphere via photosynthesis and transfer it into biomass and, more importantly, into underlying sediments where it can be stored over long timescales. The service is primarily realised through the ongoing accumulation of soil organic carbon, which represents a net removal of CO_2 from the atmosphere, as well as through the protection of existing carbon stocks that would otherwise be released if the ecosystem were degraded or disturbed. In some cases, this benefit is partially offset by emissions of other greenhouse gases such as methane, so the net climate regulation service reflects the balance between carbon sequestration and these emissions.

Soil organic carbon accumulation rates for New Zealand mangroves, saltmarsh and seagrass are sourced from Berthelsen et al., (2025). Median values were used as the central estimates because the underlying data are skewed and highly variable. Low and high sensitivity values were defined using the New Zealand dataset mean ± 1.96×SE to reflect the spread of the available evidence. These bounds are used as scenario values rather than formal confidence intervals around the median.

Table 9: Blue carbon sequestration rates. Source: Bertelsen et al., 2025

	Soil organic carbon accumulation rate (t/ha/yr)			CO2 sequestration (t/ha/yr)		
Habitat	Median	Mean	SE	Central	Low	High
Mangrove	0.39	0.64	0.20	1.43	0.91	3.78
Saltmarsh	0.59	1.08	0.19	2.16	1.90	5.33
Seagrass	0.04	0.04	0.01	0.15	0.07	0.22

The sequestration rates are applied as constant annual rates for current ecosystem extent. They represent empirical average soil carbon accumulation rates measured over multi-decadal timescales rather than asymptotic carbon stock accumulation functions. Therefore, the indicator estimates current annual climate regulation benefits and does not model potential long-term changes in sequestration rates associated with habitat maturity, changing sedimentation, sea-level rise, or other ecological dynamics.

Monetisation

Consistent with the land-based climate regulation layer, the New Zealand Treasury shadow price of carbon is applied. The estimated value for 2026 is NZ$106 per tonne of CO_2-equivalent.

Data

The marine ecosystem extents are sourced from “Our Estuaries” Seasketch layers published by DOC (https://seasketch.doc.govt.nz/arcgis/rest/services/National/Our_Estuaries/MapServer). The following layers are used:

- Estuary plants and animals - mangroves
- Estuary plants and animals - seagrass
- Marine habitats, type = “Biogenic Saltmarsh”

Steps

- Convert mangrove and seagrass density descriptions to a number. Assume medium if density is not populated

```
CASE
 WHEN "Density" LIKE 'High%' THEN 0.75
 WHEN "Density" LIKE 'Med%' THEN 0.3
 WHEN "Density" LIKE 'Low%' THEN 0.1
 WHEN "Density" LIKE 'None%' THEN 0
 ELSE 0.3 END
```

- Rasterise mangrove and seagrass densities
- Rasterise saltmarsh using a burn value of 1 (no density information)
- Run the python script “Blue_carbon_sum_sequestration.py” to apply the central, low and high rates to each cell, and multiply by the carbon price

Limitations

This indicator is based on nationally consistent habitat extent data and average sequestration rates derived from a limited number of field studies. It assumes uniform sequestration rates within each habitat type and does not account for local variation in sedimentation, hydrology, ecosystem condition, nutrient availability, or species composition. Habitat density is also represented using simplified classes for mangroves and seagrass and no density information is available for saltmarsh.

Future improvements

The indicator could potential be improved by incorporating sediment type, elevation, tidal inundation, and hydrological setting, as these influence carbon accumulation. A further improvement would be to include net greenhouse gas effects where methane or nitrous oxide emissions are material, rather than valuing soil carbon accumulation alone.

5.13. Water treatment in wetlands

Explanation

The water treatment ecosystem service provided by wetlands refers to their ability to improve water quality by removing nutrients, sediments, and microbial contaminants from water as it passes through them. This occurs through a combination of physical settling, filtration by vegetation, and biogeochemical processes such as denitrification and microbial die-off, resulting in reduced pollutant loads delivered downstream to rivers, lakes, and human receptors. While models such as InVEST NDR model and InVEST SDR model capture retention across terrestrial landscapes, they primarily represent hillslope processes and do not explicitly model in-stream or in-wetland treatment. They do not capture the additional attenuation that occurs once contaminants enter aquatic systems. As a result, wetland water treatment is treated as a distinct ecosystem service, representing downstream processing and removal beyond the scope of standard land-based retention models.

For accounting purposes, the nutrient reduction component can be added to the nutrient regulation service, and sediment and pathogen reduction values can be added to the water purification service.

Treatment capacity

To avoid overstating the treatment potential of wetlands beside rivers, only wetlands with a plausible hydrological connection to low-order streams were assumed to treat stream loads. The model therefore includes only intersecting stream segments with stream order 1 or 2.

Loads from higher-order streams are excluded because larger channels are less likely to be meaningfully intercepted or treated by adjacent wetland area.

Bogs are also excluded from stream-load treatment. This reflects their hydrological character: bogs are primarily rain-fed systems, with water and nutrient inputs dominated by direct precipitation rather than river or groundwater inflows. In the model, bog wetlands can still be represented spatially as wetlands, but they are assigned no treatment of stream-derived TN, TP, TSS, or E. coli loads. Treatment of surface flows by bogs is still represented in the general nutrient regulation indicator.

Performance

Removal rates for nitrogen (N), phosphorus (P), and sediment (S) are sourced from the New Zealand Wetland Practitioner Guide (C. C. Tanner et al., 2022). These rates vary by climate (warm versus cool) and are scaled according to the proportion of the contributing catchment in wetland, up to a maximum of 5%. This approach provides a practical representation of wetland treatment performance, but is a simplification of more complex processes that depend on hydrology, residence time, and wetland condition.

Wetlands can also substantially reduce faecal indicator bacteria through sedimentation, filtration, and die-off processes (C. C. Tanner et al., 2022). Constructed wetland studies report high removal efficiencies, typically 85-99% (Vymazal, 2005), although natural wetlands are likely to be less efficient due to lower hydraulic control. New Zealand evidence shows performance varies with residence time and connectivity (Wilcock et al., 2012). Wetlands may also act as sources under some conditions, particularly during high flows (Stott et al., 2023). Reflecting this variability, E. coli attenuation is represented using a range of 0-90%, with a central estimate of 70%.

Calculation of treatment

Wetland treatment was estimated by first delineating wetland extent using the WONI layer and identifying REC2 riverlines that intersect each wetland. Wetlands with no intersecting riverlines were assumed to interact only with diffuse surface runoff rather than channelised flow, and therefore their treatment effects are already captured in ecosystem service layers derived from the InVEST NDR and SDR models.

Nutrient and sediment inputs were derived from the riverlines immediately upstream of each wetland, with headwater catchments using their own in-stream concentrations. Nutrient inputs were based on modelled water quality from RiverMaps, with total nitrogen (TN) and total phosphorus (TP) concentrations converted to in-water loads using median flow, while suspended sediment loads (t/year) were taken directly from RiverMaps. To avoid overstating treatment potential, treated flow was capped at 0.0056 m^3/s per hectare, as calculated above. Because observed water quality already reflects attenuation by existing wetlands, estimated loads were adjusted by dividing by (1 - removal rate) for each contaminant to approximate counterfactual conditions without wetland treatment.

For microbial contamination, median, G540 and Q95 E. coli concentrations were extracted from the riverline(s) immediately upstream of each wetland. Wetland treatment was assumed to shift the distribution of E. coli concentrations without altering its underlying shape, consistent with a lognormal-type distribution. Using the same removal rates applied to nutrients and sediment, recreation attribute bands were estimated for both with-wetland and without-wetland scenarios.

Monetisation

Abatement cost-based values were used in place of downstream benefit estimates on the basis that contaminant reductions are effectively required to meet environmental objectives, and are therefore valuable at the margin. This approach avoids the uncertainty and data demands of modelling the full pathway from loads to ecological and human outcomes. While it implicitly assumes all reductions are beneficial, this is reasonable here, as the largest wetlands tend to be located in high-load catchments such as the Piako and lower Waikato. Abatement costs therefore provide a pragmatic and policy-relevant proxy for water quality benefits.

Marginal monetary values for nitrogen, phosphorus and sediment were drawn from Daigneault et al. (2017), which estimate the economic impacts of load reductions using an integrated modelling framework linking land-use change, mitigation actions, and farm profitability. These values (Table 10) therefore reflect the marginal economic consequences of reducing contaminants, including changes in producer surplus, rather than purely willingness-to-pay measures. The least-cost pathway in their framework includes on-farm measures such as partial retirement of agricultural land, which tends to yield lower marginal values than those derived from interceptive mitigation-focused modelling such as Matthews et al. (2024). The other issue with using marginal costs from Matthews et al. (2024) is that this study includes constructed wetlands as a mitigation option. Using them to value the services provided by natural wetlands would introduce circularity, as the value of the ecosystem service would be determined in part by the cost of an engineered analogue. By instead adopting externally derived values, the analysis avoids this circularity. However, alternative values are a similar order of magnitude.

Table 10: Monetary values of contaminant reductions expressed in 2026 NZD, based on inflation-adjusted estimates from Daigneault et al. (2017).

Unit	Medium	Low	High
Total nitrogen (kg)	27.2	13.6	54.4
Total phosphorus (kg)	136	68	272
Sediment (t)	4.08	2.04	8.16

E. coli is treated differently, as its impacts relate directly to human health risk from exposure, so values are based on estimated infection probabilities rather than per-unit reductions in contaminant loads. The monetary values of E. coli reductions was based on willingness-to-pay (WTP) estimates for improved recreational water quality reported in Tait (2016), adjusted to current prices using inflation and scaled to the 2023 adult population. Changes in water quality were expressed in terms of movements between bands for human recreation as defined by the 2020 National Policy Statement for Freshwater Management (New Zealand Ministry for the Environment, 2024, Table 9),

To translate these values to a spatially explicit unit, each 1 ha cell was treated as representing a fixed share of national freshwater resources (1/2644690 total hectares in all catchments). As the Tait WTP estimates are defined per 1% improvement in water quality, this share was divided by 1% to obtain the proportional contribution of each cell. The resulting factor was then multiplied by the relevant band-specific WTP value, yielding an estimate of the monetary value of E. coli improvements attributable to each hectare.

Table 11: Monetary value associated change in catchment condition between human recreation bands for 1 hectare, relative to baseline state E.

Attribute Band	Central	Low	High
A	69.04	58.19	79.89
B	23.99	13.56	34.42
C	14.60	4.59	26.70
D, E	0	0	0

Data

- Revised extent of wetlands in New Zealand (https://datastore.landcareresearch.co.nz/dataset/revised-extent-of-wetlands-in-new-zealand)
- River Environment Classification (REC2) New Zealand (https://data-niwa.opendata.arcgis.com/maps/3a4b6cc2c1c74fbb8ddbe25df28e410c/about)
- Rivermaps modelled water quality (https://shiny.niwa.co.nz/nzrivermaps/)
- nzenvds-mean-annual-temperature-v10

Steps

- Do a raster calculation on nzenvds-mean-annual-temperature to create a warm-climate dummy raster and save as warm_climate_dummy.tif:

```
calc="A>=12"
```

- Add a unique ID field to WONI features and rasterise as wetland_id.tif
- In QGIS, use “Join attributes by location” on WONI and riverlines to determine which streams go through wetlands (this is many-to-many relationship). Include the fields wetland_id, wetland_area_ha from WONI and nzsegment and catchment_ha from REC2. Save as “wetland_riverline_intersections.csv”
- Run “get_upstream_segments.py” to get the flow-weighted contaminant concentrations and loads entering each riverline. For headwaters it returns own loads and concentrations. This creates “upstream_quality_nzsegment.csv”
- Run wetland_treatment_calculation.py to calculate removal quantities
- Run wetland_treatment_calculate_value.py to calculate values

Note: For narrow or fragmented wetlands, the mapped wetland area can exceed the number of 1 hectare grid cells used to represent it. As a result, the treated flow assigned to an individual cell may exceed the nominal per-hectare maximum of 0.0056 m3/s. This does not indicate an error. It reflects the method used to distribute wetland-level totals across a discrete grid.

Limitations

This indicator uses a simplified representation of wetland treatment processes based partly on removal efficiencies derived from studies of constructed wetlands. Although treatment is restricted to wetlands with plausible hydrological connectivity to low-order streams, the approach does not explicitly model residence time, hydraulic loading, flow pathways, wetland condition, or seasonal variation, all of which influence treatment performance. The indicator also relies on modelled water quality data.

Potential improvements

A useful future improvement would be to calibrate the indicator using monitoring data from natural New Zealand wetlands. This could involve measuring inflow and outflow concentrations and flows for TN, TP, suspended sediment and E. coli across a representative sample of wetland types, catchment settings, stream orders and climates. These data could be used to estimate treatment rates directly for natural wetlands, test the current assumptions about hydrological connectivity and low-order streams, and identify which wetland characteristics best predict treatment performance.

6. Provisioning services

6.1. Crops

The crop provisioning ecosystem service represents the potential of land to produce marketable food and fibre crops, reflecting the biophysical capacity of soils, climate, and terrain rather than current land uses or production systems. Food crops can be grown for either human or livestock consumption, and some crops produce both food and fibre.

The potential supply indicator represents an upper-bound estimate of the potential economic value of arable and horticulture crop production that could be supplied by the land if allocated to the most valuable crop. Potential yields for 18 crops were sourced from landuseopportunities.nz and expressed as relative yield surfaces at a one-hectare resolution. These yield surfaces were combined with representative output values per tonne, sourced from Fresh Facts 2025, MPI vineyard monitoring reports, and the AIMI New Zealand survey of cereals 2025. For each grid cell, crop-specific potential yield was multiplied by its corresponding output value to estimate potential gross output, and the maximum value across all crops was selected as the indicator of crop provisioning supply.

The realised crop provisioning value represents current production rather than biophysical potential, and is derived directly from existing land use. It is calculated by applying typical per-hectare output values to areas classified as cropping and horticulture in the NZLUM land use dataset.

Only a proportion of realised production value is attributed to ecosystem services, recognising that observed crop returns also depend on labour, machinery, irrigation, fertiliser, infrastructure, management, and market conditions. The attribution percentages therefore represent the estimated contribution of ecosystem functions to realised production outcomes, excluding pollination where this is represented separately. Central attribution values were informed by ecosystem accounting literature using production-function approaches, which estimate that approximately 44% of crop production value can be attributed to crop provisioning ecosystem services at the EU scale, with lower and upper ranges used here to reflect uncertainty and variation between land use systems (European Commission Joint Research Centre, 2024).

Higher attribution percentages are applied to perennial and intensive horticultural systems where production is more closely linked to ecological conditions such as soil function, water regulation, microclimate, and long-term land capability. Lower attribution percentages are applied to pastoral and broadacre systems where output is more strongly mediated through managed inputs and production systems.

Table 12: Crop output value and assumed contribution from ecosystem services (JRC, 2024)

NZLUM commodity	Output $/ha	Central	Low	High
Apples	120,000	0.4	0.2	0.6
Avocados	35,000	0.4	0.2	0.6
Broccoli	30,000	0.3	0.1	0.5
Cabbages	25,000	0.3	0.1	0.5
Fodder cropping	5,500	0.2	0.05	0.35
Chamomile	12,000	0.3	0.1	0.5
Chicory	5,000	0.2	0.05	0.35
Clover	4,000	0.2	0.05	0.35
Feijoa	35,000	0.4	0.2	0.6
Flowers and foliage	50,000	0.3	0.1	0.5
Grapes	25,000	0.4	0.2	0.6
Kiwifruit	240,000	0.4	0.2	0.6
Lucerne	6,000	0.2	0.05	0.35
Maize	5,500	0.2	0.05	0.35
Melons	10,000	0.3	0.1	0.5

Olives	12,000	0.4	0.2	0.6
Persimmons	100,000	0.4	0.2	0.6
Pinenuts	15,000	0.4	0.2	0.6
Plantain	6,000	0.2	0.05	0.35
Pomegranate	30,000	0.4	0.2	0.6
Pumpkins	15,000	0.3	0.1	0.5
Tamarillo	40,000	0.4	0.2	0.6
Turnips forage	7,000	0.2	0.05	0.35
Other short rotation cropping	2,000	0.2	0.05	0.35
Other perennial horticulture	40,000	0.35	0.15	0.55
Other intensive horticulture	70,000	0.35	0.15	0.55

Limitations

The potential yield layers do not reflect the availability of water for irrigation, access to markets, labour, or the presence of supporting infrastructure such as storage and packing facilities.

Crop provisioning services are inherently co-produced, relying not only on underlying natural capital but also on substantial inputs such as irrigation, fertiliser, energy, labour, and specialised infrastructure. While the ES framework conceptually distinguishes natural capital from other forms of capital, data limitations mean this separation cannot be fully operationalised for intensive horticulture and cropping. Therefore, values reported here should not be interpreted as measures of nature's standalone contribution, but rather as indicators of gross output associated with land use and biophysical capacity. Using gross output-based indicators is nevertheless a standard and widely accepted approach in ecosystem service assessments (Burkhard et al., 2014).

Data

- Crop yield layers from https://landuseopportunities.nz/
- Iris-new-zealand-land-use-management-version-03-nzlum-v03

Steps

Create supply raster

- Select NZLUM codes from 2.3.0 to 2.5.2 (cropping and horticulture)
- Normalise all crop yields rasters to a range 0 to 1 where 1 is maximum yield for that crop
- Use the python script "Crop_potential_value.py" to calculate the crop with the maximum yield * value and create the final raster

Create realised value raster

- Select NZLUM land-use types 2.3 (seasonal cropping), 2.4 (Perennial horticulture) and 2.5 (Intensive horticulture)
- Add a field "crop_type" that uses the first commodity if available, otherwise a class default

```
coalesce( "commod_1",
 CASE   WHEN "lu_code_secondary" = 5 THEN 'Intensive_horticulture'
  WHEN "lu_code_secondary" = 4 THEN 'Perennial_horticulture'
  ELSE 'Other short rotation cropping'
 END)
```

- Join on the csv of output values by crop type

- Rasterise and export as crop_value.tif

6.2. Fodder

Explanation

The supply indicator represents pasture productivity, expressed as annual dry matter production per hectare. Pasture productivity is derived from the Pasture Production layer provided by landuseopportunities.nz, which estimates potential pasture production across all land based on climate and land characteristics, assuming no irrigation and no fertiliser inputs. This layer provides complete national coverage and is not restricted to areas spectrally consistent with higher-producing grassland, unlike the LRIS Pasture Productivity layer. Pasture production is used as a proxy for the production of other fodder crops, as spatially explicit data on the productivity of alternative forage crops are not readily available at national scale.

Realised flow indicator

The realised flow indicator reflects the portion of this pasture production that is actually utilised by livestock, and is derived by scaling the productivity estimates by an assumed pasture utilisation rate. Pasture utilisation represents the share of grown pasture that is consumed or harvested, accounting for losses due to trampling, senescence, and management constraints. Realised pasture production in New Zealand is typically around two-thirds to three-quarters of biophysical potential (Chapman et al., 2025; Edmeades & McBride, 2025). Therefore a pasture utilisation factor of 70% was adopted for scaling pasture supply to realised flow.

There is no demand indicator because there is no clear, independent measure of latent demand for pasture.

Monetisation

Pasture biomass is valued based on its contribution to livestock feed energy. Pasture dry matter in New Zealand typically contains around 8–12 MJ of metabolisable energy per kilogram of dry matter (DM)[5], so a mid-range value of 10 MJ ME/kg DM is adopted.

For dairy production systems, the DairyNZ Economic Survey reports a standard feed valuation of approximately NZ$0.39/kg DM (DairyNZ, 2025, p. 27). However, this represents a typical or benchmark value. The marginal value of pasture varies with milk price, feed constraints, and substitution with supplements. Accordingly, a broader low/high range of NZ$0.20-$0.50/kg DM ($0.02-$0.05 per MJ ME) is applied to reflect variation across seasons and production conditions. The upper range approximates the delivered cost of palm kernel expeller as a substitute feed source..

In New Zealand sheep and beef systems, pasture accounts for the vast majority of feed supply (>95%) and directly determines livestock energy intake and production (Adjabui et al., 2025). For the purposes of this assessment, fodder provision is treated as a foundational ecosystem input underpinning pastoral production. Rather than applying a formal factor-share allocation, the value of fodder is inferred from ecosystem-dependent pastoral production value, using whole-farm operating profitability as a proxy for the economic value supported by pasture production.

A spatial layer of sheep and beef operating profit for 2024 is sourced from (Harris et al., 2024) is used as a proxy for fodder value where available. For areas not covered by this layer, an average feed value is derived from Beef + Lamb New Zealand economic statistics (Beef + Lamb, 2026). Using an average EBITRm of $388 per hectare, a stocking rate of 6.2 stock units

[5] https://www.dairynz.co.nz/feed/fundamentals/feed-values/

per hectare, and a feed demand of 3,428 kg DM per hectare, this implies an average feed value of approximately $0.11 per kg DM. The low and high values are derived from observed variability in operating profit over the past five years, including the 2025-26 season forecast, and correspond to 58% and 145% of the central estimate, respectively.

The final calculation assigns pasture value only to grazing land, using modelled sheep and beef operating profit per hectare where available, and otherwise estimating value from utilised dry matter by applying a dairy-specific value of $0.39/kg DM or a non-dairy grazing value of $0.11/kg DM.

Data

- lris-lcdb-v60-land-cover-database-version-60-mainland-new-zealand
- Pasture production - No irrigation, no fertiliser N (https://landuseopportunities.nz/dataset/pasture-production-maps)
- Sheep and beef operating profit 2024 (https://landuseopportunities.nz/dataset/sheep-and-beef-economic-indicators)

Steps

- Reproject pasture production layer and save as fodder_supply.tif

Create realised flow raster

- Multiply supply x grazing land x 70% pasture utilization rate
- Export as fodder_kg.tif

Create value raster

- Create a binary raster for dairy land:
  - Select NZLUM features where "commod" like "dairy"
  - Rasterise with a burn value 1
  - Save as dairy_grazing.tif
- Create a binary raster for non-dairy grazing:
  - Select NZLUM features where lu_code begins with 2.2 (Grazing modified pasture systems) and "commod" NOT like "dairy"
  - Rasterise with a burn value 1
  - Save as other_grazing.tif
- Perform the following raster calculation:

```
where(( @dairy_grazing==1) | (@other_grazing==1), where(@snb_profit>0, @snb_profit,
    where( @dairy_grazing ==1, @fodder*0.39, @fodder*0.16)),0)
```

- Export as fodder_flow_value.tif

Limitations

A limitation of the pasture production layer used in this analysis is that it represents biophysical pasture growth potential under climate and land constraints, assuming no fertiliser or irrigation inputs. As a result, the estimates do not reflect realised pasture production under typical farm management and may understate productivity in intensively managed systems, particularly on higher Land Use Capability (LUC) land. The layer is therefore best interpreted as a measure of potential pasture supply rather than observed production.

6.3. Timber

Explanation

The Timber ecosystem service represents the capacity of plantation forests to produce wood for material and energy uses. It reflects the biological productivity of forest land, with potential supply indicated by site productivity and realised flow occurring when timber is harvested and enters economic use.

The supply indicator uses spatial variation in radiata pine volume productivity (300 Index) to represent the biophysical potential for timber production. The 300 Index measures the mean annual volume increment at age 30, normalised to a standard stand density of 300 stems per hectare, and is expressed in units of m^3/ha/yr.

The realised flow indicator is derived by applying the NZLUM land use classification to restrict the assessment to plantation forestry. NZLUM includes classes intended to distinguish permanent carbon forests, but these are not yet populated, meaning the indicator is likely to overstate timber supply by implicitly including areas not managed for harvest. NZLUM also does not distinguish the small areas of other planted species such as Douglas Fir. The approach therefore uses radiata pine productivity as a proxy for plantation forestry more generally.

Annual timber provisioning value was estimated by converting harvest-age stumpage to an annual flow over the rotation period. This is consistent with ecosystem accounting guidance that treats wood provisioning as a harvested biomass flow, and with Ghasemi et al. (2025), who estimate timber production as annual wood productivity by dividing harvestable yield by harvestable age.

Biomass energy derived from forest residues can provide a valuable supplementary energy source. However, it is not valued separately in this assessment, as residue markets are already incorporated into estimated timber harvest values, and separate valuation would risk double counting.

Monetisation

Regional radiata pine net stumpage values were taken from Manley (2021, Table 5). The central estimate uses the average-input stumpage value for each region, while the low and high estimates are based on the 25th and 75th percentile values. These values were inflated to 2026 prices and converted to an annualised timber provisioning value by dividing by the assumed 28-year rotation period. The resulting annual value per hectare was then divided by the regional average 300 Index reported in Manley (2021, Table 2) to derive an annual value per unit of 300 Index. This adjustment allows the monetary value applied to the productivity raster to vary by region, reflecting regional differences in harvest costs, slope, distance to road, and transport distance as well as biological productivity. The resulting annual values per unit of 300 Index are reported in Table 13. Finally, each cell in the 300 Index raster was multiplied by the relevant regional value to produce the spatial layer of annual timber provisioning value.

Table 13: Annual value of timber accumulation per unit of 300-index

Region	Central	Low	High
Northland Region	39.90	25.84	60.70
Auckland Region	39.90	25.84	60.70
Waikato Region	33.34	23.18	51.22
Bay of Plenty Region	47.59	35.66	64.16
Gisborne Region	46.67	40.60	61.72
Hawke's Bay Region	37.91	24.04	56.59
Taranaki Region	37.91	24.04	56.59
Manawatu-Whanganui Region	37.91	24.04	56.59
Wellington Region	37.91	24.04	56.59
West Coast Region	38.21	25.90	76.06
Canterbury Region	38.21	25.90	76.06

Otago Region	39.49	27.82	59.92
Southland Region	38.19	15.38	65.48
Tasman Region	43.17	33.48	61.88
Nelson Region	43.17	33.48	61.88
Marlborough Region	43.17	33.48	61.88

Data

- spatial-variation-in-radiata-pine-volume-productivity (https://scion.koordinates.com/layer/107990-spatial-variation-in-radiata-pine-volume-productivity/)
- new-zealand-land-use-management-version-03-nzlum-v03

Steps

Create potential supply indicator

- Reproject Pradiata_300Index to match project extent and grid
- Normalise to a range 0 to 100 and export as timber_supply.tiff

Create realised value indicator

- Select NZLUM lu_code 2.1 and zone = "Rural" or null.
- Rasterize NZLUM plantation forest with a burn-in value of 1
- Reproject 300-index to project grid
- Rasterize region code
- Create a regional lookup CSV with central, low and high values, and average 300-index to use where the 300-index has no data.
- Run the Python script *Timber_calculate_value.py* to multiply 300-index within forest extents by the region-specific value.

Limitations

This indicator uses radiata pine productivity as a proxy for all plantation forestry and does not account for differences in species, age class, management regime, or harvesting constraints. It also assumes a standard 28-year rotation and may overstate realised timber production because NZLUM does not currently distinguish permanent carbon forests from production forests. Regional stumpage values capture broad differences in profitability but not local variation in costs or accessibility.

Potential improvements

Future improvements could incorporate information on forest age classes, species composition, and management objectives to better distinguish production forests from permanent carbon forests. Linking the indicator to forestry inventory datasets and spatial harvest forecasts would also provide a more accurate representation of realised timber production.

6.4. Firewood

Explanation

The wood fuel ecosystem service is the provision of woody biomass that can be used as a source of space heating energy for households. This service arises from trees and woody vegetation across a range of land covers, including forests, shelterbelts, and riparian plantings, and contributes to household welfare by supplying a locally available source of useful heat.

Tree canopy cover data were used to indicate potential fuelwood supply, as fuelwood availability depends on the presence and extent of woody biomass rather than the dominant economic use of land. Unlike land-use classifications such as LCDB or NZLUM, tree cover directly captures the spatial distribution of trees, including shelterbelts, riparian plantings, and scattered woody vegetation within agricultural landscapes that are not separately identified in land-use datasets but can contribute to residential fuelwood supply.

The demand for wood fuel was estimated using household energy use and population data. Census 2023 was used to identify the number of dwellings using wood fuel in each SA1, while the Energy End Use Database (EEUD) reports total residential wood energy consumption of 7,442 TJ for space and water heating in 2023.

The total wood energy consumption is multiplied by 0.7 to reflect typical wood burner efficiencies (Seymour, 2020). Dividing usable energy by the number of wood-using dwellings yields an average of 10.9 GJ per dwelling per year. This per-dwelling energy use was allocated to SA1s based on the number of wood-using dwellings, summed to obtain total wood energy use per SA1, and then normalised by SA1 land area (hectares) to derive a spatial indicator of realised wood fuel use per hectare.

The realised flow of the wood fuel ecosystem service was estimated by spatially allocating observed household wood energy use to areas of woody biomass, subject to accessibility and physical constraints. To reflect the localised nature of firewood supply, a proximity weighting was applied using an exponential distance-decay function centred on wood-burning demand, calibrated so that approximately half of the influence occurred within 30 km. Realised flow was then allocated to supply cells in proportion to the product of tree cover and proximity weight, while imposing a conservative physical cap equivalent to a maximum of 1 m^3 of firewood per hectare per year, scaled by tree cover. Any excess demand beyond local capacity was iteratively redistributed to other eligible cells, ensuring that total allocated flow matched observed national consumption while remaining within plausible local supply limits.

Monetisation

The monetary value of the wood fuel ecosystem service could be anchored to market value or the cost of an equivalent unit of useful heat from another source (i.e. from a heat pump). However, the two approaches yield equivalent values in this situation.

Typical dry stacked firewood has a heat output of 1091 to 1860 kWh per cubic metre, depending on the timber species and accounting for burner efficiency (https://www.nzffa.org.nz/farm-forestry-model/tree-grower-articles/february-2019). Firewood prices were sourced from various retailers including www.firewoodfactory.co.nz, www.thefirewoodcompany.co.nz/firewood, and www.uppercutz.co.nz/. Table 14 shows that cost per GJ is in the range of $29-$60

Table 14: Typical firewood energy values (www.nzffa.org) and prices (various)

Species	**kWh/m³**	**Typical price/m³**	**Cost/kWh**	**Cost/GJ**
Poplar	1200	125	0.10	29
Eucalypt	1270	200	0.16	44
Pine	1091	140	0.13	36
Macrocarpa	1150	250	0.22	60
Manuka	1860	300	0.16	45

For a heat pump, an electricity price of NZD 0.30 per kWh and a coefficient of performance (COP) of 3 implies a cost of approximately NZD 28 per GJ of useful heat (0.30 × 277.8 / 3). As this is lower than the cost of purchasing firewood, the heat pump cost is used as a proxy to value the firewood provisioning service. Sensitivity analysis is based on variation in COP, with a higher COP of 4.5 giving a low estimate of $15/GJ, and a lower COP of 2.5 giving a high estimate of $39/GJ.

Limitations

This analysis has several limitations that should be considered when interpreting the results. Tree canopy cover is used as a proxy for potential fuelwood supply, but the presence of tree cover does not necessarily indicate that firewood can be harvested in practice. Legal restrictions, land ownership, management objectives and terrain considerations may limit access to woody biomass. In addition, tree cover does not distinguish between standing biomass, deadwood, or species suitable for firewood. The spatial weighting approach further assumes that proximity between supply and demand increases the likelihood of fuelwood use, but this does not necessarily reflect actual collection patterns, which may be influenced by informal arrangements, commercial supply chains, or personal preferences that are not spatially explicit in the data. The flow indicator should be interpreted as an indicative spatial allocation of observed fuelwood use rather than a precise representation of where firewood is harvested in reality.

Steps

Create supply indicator

- Reproject tree cover raster and mask to land area.
- Export as firewood_supply.tif

Create demand indicator:

- Add a decimal calculated field to census table for wood-using households per hectare

```
if(to_int("VAR_3_114")>0,to_int("VAR_3_114"),0)/($area/10000)
```

- Rasterise this field and save as firewood_dwellings.tif
- Multiply by 15.56GJ and export as firewood_demand_GJ.tif
- Normalise to a range 0 to 100 and export as firewood_demand.tif

Create flow indicator

- Use the Python script “firewood_distance_weighting.py” to spatially allocate demand to local tree cover, with the output being firewood_flow_gj.tif
- Multiply by $28/GJ to create firewood_flow_value.tif

6.5. Wild seafood

Explanation

The wild seafood ecosystem service represents the provisioning of wild marine biomass for commercial use. It includes all commercially harvested species recorded by MPI Fisheries, encompassing finfish, sharks and rays, crustaceans, molluscs, and marine plants landed for sale. The indicator reflects the annual economic value generated from these wild harvests and therefore captures the realised provisioning flow from marine ecosystems to the economy. Recreational catch is excluded from this indicator and is treated separately, recognising that commercial and recreational fisheries differ in reporting systems, management structures, and value pathways.

The potential supply indicator represents the relative ecological capacity of each seafloor community to support commercially harvested species. Using the DOC Seafloor Community Classification (Stephenson et al., 2020), mean taxa occurrence of all MPI-recorded commercial species was summed within each class. The resulting index was standardised to provide a relative measure of habitat-linked capacity to generate harvestable biomass, independent of current fishing activity.

Prices were then used to weight the relative commercial importance of different taxa. Either port prices or Annual Catch Entitlement (ACE) prices could be used for this purpose, but they imply different interpretations. Port prices provide a gross landed-value measure and therefore include returns to fishing effort, vessels, labour, fuel, processing, quota access, and other human or produced capital. ACE prices would provide a closer proxy for the annual scarcity rent associated with access to QMS stocks, and therefore may better approximate the ecosystem-service contribution. However, ACE prices were not freely available for this assessment. Indicative port-equivalent prices were therefore applied, using FishServe port prices where available and proxy values otherwise. Proxy values were $8/kg for premium finfish (e.g. bass, hāpuku), $3/kg for non-premium finfish, $2.5/kg for sharks and dogfish, $15/kg for crustaceans, $4/kg for molluscs, and $0.5/kg for seaweed.

Monetisation

For each species, annual landed value was calculated as port price multiplied by reported 2025 commercial catch (MPI Fisheries), using proxy values where necessary. This national species-specific value was then spatially distributed across seafloor classes in proportion to the relative mean taxa occurrence of that species within each class and weighted by the area of each class to derive a per-hectare value. Unlike the potential supply indicator, which represents ecological capacity, the realised flow captures the economic value of biomass actually harvested and allocates it to habitats based on ecological association rather than fishing effort data.

The low and high range was derived from observed variability in fisheries economic value added reported in Stats NZ's marine economy account. Species-specific variability in port prices was not available, so this sector-level series was used as a pragmatic proxy for uncertainty in realised fisheries value, capturing annual variation in prices, catch composition, costs, and productivity. In inflation-adjusted terms, the lowest year between 2007 and 2023 was 69% of the period mean, while the highest year was 125% of the mean. These factors were applied to the central estimate to indicate plausible variation in realised economic value.

Data

- New Zealand Seafloor Community Classification
- MPI Fisheries catch data
- Fishserve port prices

Steps

- Merge the two seafloor layers (exclusive economic zone and territorial sea)
- Reproject to EPSG: 2193 and an extent rounded to the nearest hundred meters so it will align with other ES layers
- Calculate the number of hectares in each class
- Match taxa scientific names to commercial species
- Multiply mean taxa by class area for each species
- Multiply port price by annual catch to get harvest value by species
- Calculate species value per area * mean taxa weight
- Sum weighted values for each class (flow value)
- Calculate sum of port price * mean taxa weight for each class and normalise to 0-100 range (supply)
- Save a csv with the class, per-hectare flow value, normalised flow, and normalized supply
- Run the python script *Wild_seafood_allocated_value.py* to create the raster layers aligned with the seafloor class layer.

Limitations

The seafloor community classification is derived from compiled survey data collected using multiple sampling gears and survey designs (Stephenson et al., 2020) which introduces potential biases related to gear selectivity, spatial sampling intensity, and uneven survey coverage. Mean taxa values may reflect detectability and sampling effort as well as true ecological abundance.

The indicators should be interpreted as relative spatial representations of ecosystem-linked provisioning capacity and realised value, rather than precise measures of fishery activity or stock biomass. In particular, the mean taxa data do not capture the ecological, cultural, biodiversity, nursery, spillover, non-use, or insurance values associated with marine reserves and other protected areas. These areas may contribute to wider fishery productivity through larval export and spillover effects, even where extractive harvest value within the reserve itself is low or absent.

Future improvements

Future improvements could strengthen both the ecological realism and economic interpretation of the estimates. However, these refinements would require access to species-level stock assessment outputs, spatial biomass estimates, catch-effort data, ACE price data, and potentially confidential fisheries datasets that are not freely available. With those data, the approach could incorporate species-specific habitat suitability, stock status, trophic relationships, larval dispersal, spillover from marine reserves, and resource-rent estimates based on ACE prices rather than port prices.

6.6. Aquaculture

Explanation

In-water aquaculture, particularly mussel farming, is highly dependent on ecosystem services because production relies directly on natural marine conditions rather than manufactured feed inputs. Mussels obtain food by filtering naturally occurring phytoplankton from the water column and therefore depend on ecosystem processes such as primary productivity, water quality, nutrient cycling, currents, and suitable sheltered growing conditions. The New Zealand industry is also reported to be heavily reliant on wild mussel spat (Alfaro et al., 2011). Farmed King salmon, on the other hand, are fed manufactured feed and rely heavily on infrastructure, husbandry, breeding, veterinary management, and operational inputs (Iversen et al., 2020). However, open-net salmon farming also depends materially on regulating ecosystem services, including oxygen supply, water circulation and flushing, waste assimilation, water purification, disease regulation, and suitable temperature conditions.

The proportion of mussel, oyster, or salmon aquaculture revenue attributable to ecosystem services is uncertain and no published estimates could be found. A central attribution value of 60% was applied, with a low value of 40% and a high value of 80% used for sensitivity analysis. These ranges are judgement-based assumptions informed by the ecological dependence of in-water aquaculture systems described in the literature, rather than empirically estimated attribution coefficients.

Monetisation

The realised flow value for aquaculture was calculated by multiplying total production volumes of salmon, oysters and mussels, sourced from the MPI Aquaculture Performance Annual Report 2024, by corresponding port prices. Port prices for mussels and oysters were obtained from Fishserve. In the absence of a consistent published port price series for farmed Atlantic salmon, a representative port price of NZ$10 per kilogram was adopted. This value sits within the range of high-value finfish species recorded in commercial catch statistics and provides a conservative benchmark for modelling purposes. For example, the port price data indicate values of approximately NZ$10 per kilogram for bigeye tuna and around NZ$9 per kilogram for red snapper.

Total value was calculated as tonnes multiplied by port price and then spatially allocated to individual marine farms based on structure codes, with longlines assumed to represent mussel production, longlines with cages assigned to oysters, and finfish cages assigned to salmon.

A 5 km radius was adopted to represent the broader marine environment that supports aquaculture production beyond the immediate farm footprint. This distance reflects the spatial scale over which coastal hydrodynamic processes such as tidal exchange, mixing and plankton transport typically operate within sheltered embayments and coastal channels in New Zealand. Mussel production, in particular, depends on phytoplankton supply delivered by local circulation rather than conditions strictly within the farm boundary, while finfish aquaculture relies on water exchange and dilution processes operating at comparable coastal scales. Each marine farm's value was distributed across surrounding ocean cells within the 5 km radius using an exponential distance-decay function:

$$w(d)=\exp(-d/\lambda),$$

where d is distance from the farm and λ controls the rate of decay. Any value that would have overlapped land was redistributed to ocean cells.

Data

- MPI MARINE_Aquaculture_Marine_Farms
- Fishserve port prices

Steps

- Add a calculated field for value_per_ha to differentiate the species

```
CASE
 WHEN lower("Structure_type") = 'finfish cages' THEN 156527
 WHEN lower("Structure_type") IN (
    'intertidal racks',
    'longlines and cages',
    'longlines with baskets/cages'
   ) THEN 5760
 ELSE 642 END
```

- Add a calculated field value = marine farm area ha * value_per_ha
- Export as a shapefile of centrepoints and associated values
- Run the Python script "Marine_farms_allocate_value.py" to allocate value using the distance decay function. This creates the layer aquaculture_flow.py
- Normalise to a range 0 to 100 to create the layer aquaculture_flow.py

Limitations

Production volumes and representative port prices are applied at a national level, meaning the approach does not capture variation in productivity, operating conditions or prices between individual farms or regions. In addition, the use of a uniform 5 km exponential distance-decay function is a simplified representation of hydrodynamic processes and does not explicitly model site-specific circulation, nutrient delivery or waste assimilation. The mapped values should therefore be interpreted as an indicative spatial representation of current realised economic flows rather than a precise quantification of local ecosystem contribution.

Future improvements

Future improvements could involve developing a national aquaculture potential supply layer by combining the DOC Seafloor Community Classification with wave exposure, tidal current and nutrient or chlorophyll layers. The seafloor classification provides a structured ecological baseline, while hydrodynamic and water quality data would allow more defensible identification of physically and biologically suitable sites. Some of the necessary wave and circulation modelling may already exist within regional councils or other coastal planning agencies, and integration of these datasets could support a more robust potential supply assessment in future work.

6.7. Freshwater

Explanation

The water provisioning ecosystem service refers to the contribution of ecosystems to the supply of freshwater for human use. This includes water that is generated through natural hydrological processes, such as precipitation, infiltration, storage and runoff, and subsequently becomes available for abstraction for uses such as irrigation, municipal supply, industry and energy generation.

Water provisioning supply is represented by modelled annual water yield, which reflects spatial variation in landscape-generated runoff (Ausseil et al., 2013; Dymond et al., 2012). In regions where groundwater abstraction or inter-catchment transfers occur, water use may exceed locally generated yield. This reflects the influence of upstream contributions, aquifer storage and regulatory infrastructure rather than an inconsistency in the indicator. For the purposes of this assessment, water yield is used as a spatial proxy for potential supply within catchments.

Realised flow

The flow layer represents the use of surface water within each sea-draining catchment. Water use was derived from consented freshwater takes and modelled irrigation and hydroelectricity generation demand. For urban and industrial uses, the maximum annual consented volume was used, as data on actual abstraction volumes were not available. This approach may overstate realised use, as consented volumes represent allocation ceilings rather than observed withdrawals. For irrigation, surface water demand estimates were sourced from Aqualinc (2021), which provide modelled annual water requirements under baseline climate and current irrigated land-use. Annual water volumes associated with hydro generation were estimated from reported 2024 electricity generation, dam heads for each station and an assumed turbine efficiency of 85 percent.

The total annual volume and value of water use were allocated spatially within each catchment using the water yield raster as proportional weights. In several catchments, total water use exceeds modelled annual water yield. This can occur because many uses return a proportion of abstracted water to the system, and hydro generation does not remove water from the catchment. It may also reflect the use of maximum consented volumes rather than actual abstraction. In addition, water may be abstracted in one part of a river system and supplied to areas beyond the local runoff-generating catchment, particularly in systems influenced by storage or inter-catchment transfers.

Monetisation

To avoid double counting, water yield is valued as the benefit derived from the use of water, while water regulation (section 5.4) was valued using the cost to store water. The two services therefore represent distinct benefit streams.

Given the absence of market prices for raw water in New Zealand, indicative values were selected to provide plausible order-of-magnitude estimates differentiated by water use category. These values should be interpreted as benefit-transfer proxies rather than direct estimates of willingness to pay or resource rent.

Urban and industrial water values were benchmarked against observed volumetric water charges but adjusted downward to reflect that retail tariffs include treatment, pumping, storage, reticulation, monitoring, compliance, renewals, and administration. Current volumetric water charges include \$2.59/m³ in Waikato District and \$3.87/m³ in Tauranga. A value of \$1.00/m³ value is therefore interpreted as a conservative proxy for the contribution of raw water yield to supplied water services.

Lower unit values are applied to mixed, other, unspecified, and stock-water uses to reflect weaker evidence on the economic purpose of the water and the likelihood that these categories include lower-value, untreated, or self-supplied uses. Mixed or other uses are assigned \$0.25/m³ because they may include some relatively high-value activities, but the absence of a clear end use makes it inappropriate to apply the urban or industrial value. Unspecified uses are assigned a slightly lower value of \$0.20/m³ to reflect greater uncertainty and the possibility that some consents relate to low-value or non-consumptive uses. Stock water is assigned \$0.05/m³ because it is generally untreated, often self-supplied, and used as an agricultural input with a lower marginal value than municipal, industrial, or irrigation water.

Irrigation water was valued at \$0.15 per cubic metre, at the conservative end of the range reported by Irrigation New Zealand (approximately \$0.15 to \$0.40 per cubic metre), as reported irrigation water prices generally include delivery infrastructure and distribution costs in addition to the underlying water value. Hydro electricity generation was valued using a productivity-based approach, applying a value of \$0.13/kWh to estimate scheme-specific water values ranging from \$0.01 to \$0.18 per cubic metre.

The low-high range was defined using use-specific sensitivity bounds (Table 15). Where external evidence was available, such as irrigation water values, the reported range was used directly. For other water uses, ranges were specified around the central indicative value to reflect uncertainty in the proxy exchange value, with wider bounds applied to mixed and unspecified uses. Hydroelectric water values were assigned the narrowest uncertainty range because the value of water is comparatively well defined by the electricity price and the physical characteristics of each scheme.

Table 15: Monetary estimates for water yield

Primary use	Value per m3 (NZD)			m³/year
	Central	Low	High	
Domestic	1.000	0.500	1.500	1,657,529,537
Hydroelectric	0.020	0.019	0.021	49,264,412
Industrial	1.000	0.500	1.500	985,648,895
Irrigation	0.150	0.150	0.400	5,379,378,303
Mixed/other	0.250	0.100	0.500	924,488,434
Not specified	0.200	0.050	0.300	137,900,403
Stock	0.050	0.020	0.100	696,543,070

Conceptually, the approach is considered suitable for national-scale ecosystem service assessment because it links water yield to realised economic uses rather than treating all water volumes as equally valuable. The use of differentiated values recognises that the benefits derived from water depend on how it is used and is therefore consistent with ecosystem accounting guidance, which emphasises that ecosystem service values should reflect the benefits generated by service flows rather than physical quantities alone (NCAVES & MAIA, 2022).

Data

- Statistics NZ consented freshwater takes
- LRIS water yield
- Transpower sites (gives hydro locations)
- EMI gen_all (generation by station)

Steps

- Select consented takes for surface water only, excluding irrigation.
- Add the following calculated field water_value:

```
“maxannual” * (CASE
WHEN "primaryuse3" = 'Domestic' THEN 1
WHEN "primaryuse3" = 'Industrial' THEN 1
WHEN "primaryuse3" = 'Mixed/other' THEN 0.25
WHEN "primaryuse3" = 'Not specified' THEN 0.2
WHEN "primaryuse3" = 'Stock' THEN 0.05
ELSE NULL END)
```

- Use “Join attributes by location (summary)” to sum maxannual and water_value by sea draining catchment
- Rasterise irrigation demand.
- Use “Join attributes by location (summary)” to sum irrigation demand by sea draining catchment
- Join hydroelectricity water use and value csv on Transpower sites
- Use “Join attributes by location (summary)” to sum hydroelectricity water volume and value by sea draining catchment
- Join the three sea draining catchment layers and add calculated fields for total volume and value:

```
coalesce("consented_volume_excl_irrigation",0) +
coalesce("irrigation_volume",0) +
coalesce("hydro_water_volume",0)
```

- Run the python *script water_yield_allocation.py* to spatially allocate catchment-level water volume and economic value to the raster grid.

Limitations

Water use estimates are derived from a combination of consented takes, modelled irrigation demand, and hydroelectric generation data, and may differ from actual annual withdrawals.

The valuation approach relies on indicative unit-value proxies that vary by water use type and are intended to represent broad orders of magnitude rather than precise market values. Water values can vary substantially depending on location, reliability, seasonal availability, infrastructure constraints, and alternative water sources. Hydroelectric values are also sensitive to electricity prices, generation efficiency, and operational conditions.

The spatial allocation of values is also approximate. Water use is distributed within each sea-draining catchment using water yield as a weighting surface, but this does not explicitly represent hydrological connectivity between individual locations and abstraction points.

Future Improvements

One priority for future improvement would be the use of observed water abstraction data rather than consented allocation volumes for municipal, industrial, and other consented uses. Consented takes represent maximum authorised volumes and may differ substantially from actual water use, introducing uncertainty into estimates of realised service flows and associated benefits.

A second priority would be the incorporation of explicit hydrological connectivity into the flow modelling. This would allow values to be linked more accurately to the areas that contribute water to downstream users and improve representation of river networks, storage infrastructure, and water transfers.

6.8. Biochemicals

Explanation

The biochemicals and medicine service refers to the provisioning of natural products from ecosystems that are used for medicinal, pharmaceutical, biochemical, or cosmetic purposes. Within ecosystem services classifications, this service captures bioactive compounds derived from plants, animals, fungi, and microorganisms that contribute to human health and wellbeing. Unlike food or fibre, the emphasis is on compounds with therapeutic or functional properties, whether used directly in traditional practices or incorporated into commercial health and cosmetic products.

In New Zealand, candidates include mānuka products derived from Leptospermum scoparium, rongoā and other native medicinal plants (e.g. kawakawa, kānuka, and harakeke), and marine bioactives sourced from coastal and marine ecosystems. Mānuka is the most commercially visible example, particularly through honey and essential oil products marketed for their antimicrobial properties. Marine bioactives include compounds derived from seaweeds and other marine organisms that may have nutraceutical or pharmaceutical applications. There is a small but genuine and growing market for products that utilise mānuka's active compounds rather than the honey itself. The principal bioactive constituents include methylglyoxal (MGO), dihydroxyacetone (DHA), and leptosperin and related phenolic compounds (Wang et al., 2024). These compounds are associated with antimicrobial activity and are used in applications such as wound dressings, medical gels, and antimicrobial coatings. However, no information could be found on the value of this emerging market to New Zealand.

Only mānuka honey has reliable and consistent commercial data available. For this reason, mono-floral mānuka honey exports are used as the quantitative indicator of the biochemicals and medicine provisioning service. The remaining candidates, including rongoā and marine bioactives, relate more to cultural identity, traditional knowledge, and non-market values rather than measurable commercial output.

Supply indicator

The potential supply of biochemicals from mānuka was modelled as an environmentally mediated suitability index constrained to areas mapped as mānuka/kānuka in the LCDB. Three environmental drivers identified as influential for nectar quantity and quality in mānuka, namely temperature, radiation, and soil moisture conditions (Sheridan, 2019), were incorporated using NZEnvDS layers: mean temperature of the warmest quarter, mean annual solar radiation, and annual water deficit. Temperature and radiation were normalised to a 0–1 range using percentile scaling (5th–95th percentiles) to reduce sensitivity to outliers. Annual water deficit was first normalised to 0–1 using its observed range and then transformed into a mid-range suitability function to reflect the assumption that both very low and very high deficits may constrain nectar quality.

Because the economic valuation is based on mānuka honey, an additional landscape dominance modifier was included to account for the requirement that nectar collected by bees be derived predominantly from mānuka rather than competing floral sources. For each 1 ha cell, the proportion of surrounding land cover classified as mānuka/kānuka was calculated within a 5 km radius, approximating the upper range of managed honey bee foraging distances. The resulting supply index was calculated as:

$$S_i = P_i T_i R_i \left(1 - \frac{D_i - d^*}{\omega}\right)$$

where S_i is the supply index for cell i, P_i is the proportion of mānuka landcover around the cell, T_i is normalised mean temperature of the warmest quarter, R_i is normalised mean solar radiation, and D_i is normalised annual water deficit. d^* and ω are the assumed optimal deficit and tolerance width and both are set to 0.5. The resulting index is scaled to 0 to 100. This index represents relative environmental suitability for mānuka nectar bioactivity potential within existing mānuka extent and does not account for genetic variation.

Realised flow

Realised flow was valued using the export value of monofloral mānuka honey, estimated at $330 million in 2023 (Apiculture New Zealand, 2024). The low and high sensitivity bounds were based on observed variability in export values over the past 10 years, and were set at 80% and 122% of the central estimate, respectively. The total value is first allocated across regions using Stats NZ regional employment counts for ANZSIC class A019300 Beekeeping as a proxy for the spatial distribution of mānuka honey production. Within each region, the allocated value was then distributed to mānuka/kānuka supply cells in proportion to their supply index values. This approach links realised flow to observed market value while using regional beekeeping activity and local environmental suitability to spatially allocate the national export value.

Data

- lris-lcdb-v60-land-cover-database-version-60-mainland-new-zealand
- nzenvds-mean-annual-solar-radiation-v10
- nzenvds-mean-temperature-of-the-warmest-quarter-v10
- nzenvds-annual-water-deficit-v10
- Statistics NZ Geographic units and employment by region and industry 2000-2024
- MPI Apiculture monitoring 2024

Steps

- Rasterise the LCDB landcover class mānuka / kānuka
- Run the Python script *Biochemical_manuka_concentration.py* to calculate cover dominance
- Run the Python script *Biochemical_manuka_supply.py* to create the supply raster

Limitations

The valuation is based solely on the export value of monofloral mānuka honey and may therefore underestimate the broader biochemicals and medicine service by excluding products such as mānuka essential oils, nutraceuticals, and other bioactive compounds. Conversely, it may overestimate the ecosystem contribution because honey export values also reflect the inputs of beekeeping, processing, branding, and other forms of human and produced capital.

The biochemical production layer for mānuka is subject to considerable uncertainty because it extrapolates relationships derived primarily from controlled experimental studies to landscape scales. Although temperature, radiation, and soil moisture have been identified as important influences on nectar production and composition, the strength and consistency of these relationships under natural field conditions remain uncertain. In particular, biochemical characteristics such as dihydroxyacetone (DHA) concentration are influenced by additional factors that are not represented in the model, including genetic variation among mānuka populations and cultivars, local solar exposure, and other site-specific environmental conditions.

7. Cultural services

The monetary values reported in this section cover only those cultural ecosystem services that can be represented within an environmental economics framework, such as recreation, aesthetic amenity, and willingness to pay for biodiversity outcomes. Tadaki et al. (2017) note that environmental values are plural and often incommensurable, with economic valuation capturing only a subset of the ways people relate to ecosystems. This limitation is particularly important for Māori systems of value, which are grounded in mātauranga Māori, whakapapa, mauri, mana, kaitiakitanga, reciprocity, collective wellbeing, and relationships between people and te taiao. These values are not simply preferences held by individuals and are not readily reducible to willingness to pay or other monetary measures. Harmsworth and Awatere (2013) therefore distinguish Māori cultural values from ecosystem-service "cultural services" and argue that Māori values extend across the whole ecosystem-services framework rather than sitting within a narrow cultural-services category.

Consistent with these limitations, the following sections focus only on those cultural ecosystem services for which biophysical indicators and monetary valuation approaches can be applied within a welfare-economic framework. The indicators cover recreation, hunting, aesthetic amenity, and natural diversity and biodiversity-related non-use values.

International visitor values are excluded because the recreation layer is intended to estimate marginal welfare effects accruing primarily to New Zealand residents. This is consistent with Treasury CBAx guidance, which frames CBA results as impacts on the collective living standards of New Zealanders. Tourism expenditure and associated GDP, employment, or regional income effects would be treated separately through economic impact analysis.

7.1. Recreation

This study uses a gravity-based model to allocate recreational demand across space, which is consistent with current ecosystem services mapping practice that combines information on site quality, accessibility and population distribution (Paracchini et al., 2014). Recreation demand is treated as exogenous and is based on survey-derived participation rates and trip-frequency assumptions. This is appropriate for estimating the current value of recreation ecosystem service flows, because the model allocates observed or estimated existing demand across available recreation opportunities. Estimating changes in recreation value under alternative ecosystem scenarios would require further consideration of endogenous demand responses, including changes in total trip numbers, destination choice, and substitution between sites or recreation types.

Like existing approaches, the model assumes that people are more likely to visit nearby sites and that higher-quality or more attractive sites draw more visitors. However, it improves on generic ES mapping methods in three key ways. First, it applies activity-specific participation rates, so different recreation types generate different levels of demand rather than treating all residents as identical users. Second, both the distance decay functions and the attractiveness surfaces are tailored to each recreation type, reflecting differences in how far people are willing to travel and what characteristics make sites appealing. Third, demand is allocated explicitly between competing sites, so increases in one site's attractiveness reduce pressure on others. This produces a clearer estimate of where recreational use is likely to occur, rather than simply mapping potential access or recreation pressure. Specific modifiers for each recreation type is described in more detail in the following sections.

Land-based participation rates by region are taken from the Active NZ Survey (Sport New Zealand, 2025), while water-based participation rates are drawn from Water Safety New Zealand (2022). Although the DOC survey reports on a wider range of outdoor activities, its results are not available by region, so the Sport NZ rates are used where regional estimates are required. Trip-frequency assumptions are informed by the Sport NZ survey, the Australian AusPlay Survey (https://www.ausport.gov.au), and the UK Active Lives Survey.

Process

The recreation valuation method is implemented through three linked Python scripts. *Recreation_calculate_demand.py* estimates the spatial distribution of recreation demand. It takes the trip value per adult by region for each recreation type and distributes that value across SA1 areas, providing the demand surface used in the flow allocation model.

Recreation_calculate_flow.py allocates this demand to supplying areas using the demand surface, recreation-specific attractiveness (supply) surface, distance-decay parameters, and road access assumptions. For each 100 m grid cell, demand is allocated to surrounding supply cells in proportion to their relative attractiveness and travel distance. Attractiveness is raised to an exponent, α, allowing the model to vary the strength of preference for higher-quality sites. Distance effects are represented using an exponential decay function, where the median travel distance defines the distance at which a site receives half the weight of an otherwise equivalent nearby site. The kernel is truncated at a specified cut-off distance to reduce computation while retaining the relevant travel catchment.

Because attractiveness is used only as a relative weight, the absolute scale does not affect the allocation. What matters is how attractive each supply cell is compared with other accessible alternatives within the travel catchment. The model uses Fast Fourier Transform (FFT) convolution to apply this spatial allocation efficiently across the full raster. Without this step, a national 1 hectare grid would involve trillions of potential demand-supply comparisons, turning a modelling exercise into a procurement case for a supercomputer.

Each recreation type has its own attractiveness surface, generated by a separate script named *recreation_[type]_attractiveness.py*. The variables used to construct attractiveness differ by recreation type and are described in the relevant subsections. FFT convolution is most appropriate where distance effects are reasonably represented as operating in all directions from each cell, so access constraints are handled by incorporating them into the recreation-specific attractiveness surface.

Short walks

ES values for non-tramping walking reflect the value of accessible green spaces that offer nearby residents opportunities for nature exposure, relaxation and everyday physical activity. The category also acts as a proxy for other informal urban recreation activities, such as leisure cycling, jogging, and casual outdoor use undertaken for enjoyment rather than for commuting. It does not include organised sports, which are assumed to receive minimal benefits from ecological qualities.

The participation rate by region is sourced from the Active NZ survey (Sport New Zealand, 2025). As the survey does not report the number of walks per participant, annual walking frequency is estimated using AusPlay 2025 (https://www.ausport.gov.au/clearinghouse/research/ausplay/results), which indicates an average of approximately 170 walks per year among walking participants.

The AusPlay average walk duration is around 85 minutes, equivalent to approximately 7.1 km at a standard walking speed of 5 km/h. While 85 minutes may appear lengthy for a single walk, it equates to around four hours of walking per week, which is broadly consistent with public health recommendations and may in practice be accumulated through a mix of activities such as dog walking and other informal recreational walking undertaken close to home.

Short walks are supplied by Reserve Land plus LCDB classes that include urban parkland, forests, and all non-productive land-based classes. Tree canopy cover was included as a modest amenity modifier for walking, increasing the base parkland attractiveness score by up to 30% where complete tree cover is present. This reflects evidence that trees are commonly identified as preferred features of urban parks and forests, and that shade and trees increase the attractiveness of walking environments. The uplift is capped at 30% because tree cover is only one contributor to walkability, alongside access, paths, maintenance, openness and perceived safety (Schroeder, 1982; Tabatabaie et al., 2023). The attractiveness score is therefore:

100 * (0.7 + 0.3 * sqrt(cover_proportion))

The median distance between a person's home and the greenspace used for a short walk is assumed to be 1.5 km. This does not represent the length of the walk itself, which is assumed to average approximately 7.1 km per trip. Rather, it reflects the tendency for recreational walking activity to be concentrated in nearby greenspaces, particularly urban reserves and local parks.

A value of NZ$4 per visit was adopted for short recreational walks in urban greenspace. This estimate is consistent with international travel cost and stated preference studies of urban park use, which typically report per-visit values in the range of NZ$3–12 (e.g. Bateman et al., 2011; UK National Ecosystem Assessment, 2011). It is also aligned with New Zealand transport appraisal guidance, where monetised benefits of walking trips (including health and amenity components) commonly fall within the range of NZ$2–6 per trip.

Walking values are limited to the greenspace-attributable health benefits of physical activity, to avoid double counting other ecosystem services provided by urban greenspace. The value is based on the Waka Kotahi active-modes technical paper, which recommends a walking health-benefit value of NZ$4.40 per km in 2018 dollars (Weerappulige & Khoo, 2020).

However, the full walking value is not attributed to ecosystem services, because some walking would occur even without access to greenspace. The model therefore applies a greenspace-attributable fraction to represent the marginal contribution of greenspace to walking demand. This is based on Lachowycz and Jones (2014), who found that the greenest areas had 13-18% more recreational walking days than the least green areas. Expressed as a share of total walking in the greener-area case, this implies attributable fractions of 13/113 = 11.5% and 18/118 = 15.3%, which are used as the low and high estimates. A central value of 12.5% is applied. This assumption is also broadly consistent with Astell-Burt et al. (2014), who found higher walking frequency in greener neighbourhoods.

Applying these attribution factors gives values per trip of $5.14, with a low estimate of $4.11 and a high estimate of $7.41, in 2026 dollars.

Data:

- LCDBv6
- Reserve land (https://data-walkingaccess.opendata.arcgis.com/maps/da65709246fe4e6eb8f8aad972cb0de7)

Tramping

Tramping recreation supply is assumed to arise primarily from areas containing DOC tracks. A 1 km buffer was created around these tracks to represent the surrounding landscapes that contribute to the overall recreation experience. While views can extend beyond this distance, a formal viewshed analysis has not previously been published and is beyond the scope of this project. The LCDB classes within the buffered area are used to estimate the attractiveness of each track segment, independent of proximity to population.

Attractiveness values were assigned on a 0-1 scale to reflect how strongly each landcover type contributes to the quality of a tramp within the 1 km buffered corridor. Indigenous forest was set at 1.0 as the benchmark tramping landscape. High scores (0.75–0.90) were given to natural and scenic environments such as rivers, coast, lakes, tussock and alpine vegetation, reflecting their contribution to wilderness character and immersion. Moderate scores were assigned to semi-natural or modified vegetation, while highly developed or intensive land uses such as built-up areas and cropland received low values.

International valuation evidence for tramping is limited, and the available studies are not always directly comparable to New Zealand conditions. Many focus on short-duration hiking or urban nature trails, whereas New Zealand tramping often involves remote backcountry settings, multi-day trips, and extensive hut and track networks. For example, Kaur et al. (2025) estimated a recreational value of CAD 19.23 (approximately NZD 24) per visit for a large urban nature park in British Columbia, which is not the same as backcountry tramping in New Zealand. Given the limited transferability of international estimates, the New Zealand-specific values reported by Kaval and Yao (2010) are considered the most appropriate evidence source despite their age.

Kaval and Yao (2010) report an average consumer surplus of $244 per person per recreation day (12 hours) for tramping/backpacking in New Zealand. The estimate is based on five observations and has a coefficient of variation of 0.88, which implies an approximate 2007 NZD range of $29 to $458 per recreation day using the mean ± one standard deviation. Adjusted for inflation, and treating a day tramp as a half-day, this results in a central estimate of $198 per trip, with a range of $24 to $373. A two-day overnight tramp corresponds to roughly 2 recreation days, or $794 ($95 to $1493) per trip.

Data:

- LCDBv6
- DOC Tracks

Mountain biking

Mountain biking generates cultural ecosystem service (ES) value through opportunities for outdoor recreation, physical activity, and experiential interaction with natural landscapes. These benefits arise from access to suitable land-based environments that support trail infrastructure and riding experiences.

In this analysis, the service is supplied by urban parks, plantation and indigenous forest, and other non-productive land-based classes that are compatible with mountain biking activity. These land-cover types provide the physical setting within which riding occurs.

Spatial attractiveness is modelled based on proximity to established mountain bike parks. Park locations were sourced from Trailforks.com and used as focal points of activity. Attractiveness declines with distance from these locations, with a sharp drop-off beyond approximately 2 km, reflecting the spatial concentration of riding activity around recognised trail networks. A per-trip recreation value of NZ$67 was applied to mountain biking visits, with a low-high sensitivity range of NZ$51-NZ$106. This figure is inflation-adjusted from Dhakal et al.(2012), who estimated a willingness-to-pay value of NZ$48 per trip, with a 95% confidence interval of NZ$36-NZ$75 in the original price year.

A median travel distance of 65 km was assumed, with a kernel cutoff at 250 km, reflecting the regional and destination-based nature of mountain biking participation, the concentration of use around established trail networks, and evidence that major New Zealand mountain biking sites attract visitors from well beyond their immediate local catchments (Yao et al., 2023).

Data

- LCDBv6
- Mountain biking park locations (Trailforks.com)

Coastal recreation

Beach visitors typically engage in multiple activities during a single trip. Rather than modelling each activity separately, this indicator represents a generic "beach visit" that captures the overall recreational experience at the coast.

An attractiveness surface for beach visits was developed within a 1 km coastal buffer. Attractiveness at each cell is determined by three components: (i) surrounding landcover, (ii) road access, and (iii) proximity to the coastline. The landcover contribution is calculated as a weighted mean of landcover scores within a moving window around each cell, using an exponential distance-decay kernel so that nearby landcover has greater influence. Road access (derived from a distance-to-road raster) and coastline proximity (distance-to-coast raster) are applied as multiplicative scalars bounded between 0 and 1. This ensures that attractive natural settings, good access, and closeness to the shoreline jointly determine overall suitability for beach recreation.

The National Coastal and Water Safety Survey (Water Safety New Zealand, 2022) reported that 83% of adults visited the New Zealand coast in the past 12 months, with an average of 3.4 visits per month and 2.1 hours per visit (excluding travel time). This corresponds to approximately 86 hours per adult per year.

There are few New Zealand beach valuation studies (Egan et al., 2023; Y. Matthews et al., 2018) so the value is also informed by consumer surplus estimates from Australian studies (Blackwell, 2007; Pascoe, 2019; Prayaga, 2017; Rolfe & Gregg, 2012). Greater weight is placed on the New Zealand evidence because of its local relevance, with each New Zealand study assigned a weight of 30% and each Australian study assigned a weight of 10%. The resulting weighted value is interpreted as a high-amenity beach estimate, as the available studies are generally based on popular coastal destinations rather than a representative sample of all beaches.

To avoid overstating national beach recreation value, a representativeness adjustment is applied before aggregation. The central estimate uses 75% of the weighted per-visit value, with sensitivity testing using 50% for the low estimate and 100% for the high estimate. This reflects uncertainty about the extent to which values from high-use study sites exceed the average value of beach visits across New Zealand, while recognising that actual visitation is itself concentrated at accessible and attractive beaches rather than distributed evenly across the coastline. The resulting per-visit values are $21.21 to $131.35, with a central estimate of $47.21.

Table 16: Beach recreation consumer surplus estimates transferred

Study	Context	Reported consumer surplus	Weight	NZD 2026 per-visit equivalent		
				Central	Low	High
Egan (2022)	Bay of Plenty Region, NZ	NZD 121.21 per day (NZD 91 to NZD 164)	30%	141.82	106.47	191.88
Matthews (2018)	Coromandel Peninsula, NZ	NZD 12.66 to 19.64	30%	21.65	16.96	26.33
Prayaga (2017)	Locals and visitors, Capricorn Coast beaches	US$16.95 per person per visit	10%	38.61		
Rolfe and Gregg (2012)	Six Queensland beach locations	A$23.79 to A$56.98 per person per beach visit	10%	69.47	40.92	98.02
Blackwell (2007)	Surf beaches in Sunshine Coast and Western Australia	A$0.50 to A$250 per person per visit, depending heavily on travel-cost specification	10%		0.99	497.15

Pascoe (2019)	Sydney beaches	A$7.51 to A$39.83 depending on activities and visitor status	10%	30.98	12.00	63.66
Weighted average				62.95	42.42	131.35
With representativeness factor adjustment				47.21	21.21	131.35

Recreational freshwater fishing

The ecosystem service provided by freshwater fishing is the recreational benefit derived from opportunities to catch fish in rivers and lakes, supported by healthy aquatic ecosystems and adequate habitat and flow conditions. Fishing was modelled separately from other freshwater activities such as swimming because it depends on different ecological drivers (e.g. species presence, habitat suitability, flow conditions), has distinct participation rates and trip frequencies, and generates different per-trip values. It is not included in the provisioning service layers because, in this context, freshwater fishing is primarily a recreational use rather than a commercial harvest; its value arises from the experience and amenity of fishing rather than the market value of fish as a food product.

Attractiveness

Fish species were assigned relative recreational desirability scores on a 0–1 scale. Scores were based on management status, legal harvest status, and prominence in recreational fisheries as documented in the NIWA Atlas of New Zealand Freshwater Fishes[6]. Species with high recreational and management significance, particularly salmon and trout, were assigned the highest scores. Lower scores were assigned to species with limited recreational targeting or restricted harvest status. These scores were used to weight habitat suitability by recreational appeal.

For lakes, recreational fishing attractiveness was represented using the Lake Submerged Plant Indicator (SPI) score. Lake SPI captures macrophyte community structure and is widely used in New Zealand as an indicator of lake ecological health, particularly reflecting nutrient enrichment and water clarity.

For rivers and streams, fishing attractiveness was calculated as the product of (i) average species desirability, (ii) scaled Macroinvertebrate Community Index (MCI), (iii) scaled water clarity, and (iv) scaled water colour class (WCC). This multiplicative structure ensures that poor performance in any single attribute proportionally reduces overall attractiveness, reflecting the ecological reality that viable recreational fisheries require adequate performance across multiple environmental dimensions.

Riverlines with a Strahler stream order below 3 were excluded. Many of these low-order mapped channels no longer exist as natural streams in practice, having been modified, diverted, or incorporated into stormwater and farm drainage networks. To further reflect practical recreational access, only riverline cells intersecting public conservation land were retained, excluding reaches located solely on private property where public access may be unavailable even if it is near a road.

Similarly to other freshwater recreation, attractiveness is moderated by distance to a public road. The median distance is set to 500m to allow for recreational anglers who are happy to walk for a while to their preferred spot. In practice, most streams and lakes are close to roads so this constraint only limits in remote areas that attract few visits anyway.

[6] https://niwa.co.nz/freshwater/nz-freshwater-fish-database/niwa-atlas-nz-freshwater-fishes

Demand

Regional participation rates were sourced from the 2022 Water Safety Survey (Water Safety New Zealand, 2022), which reports land-based fishing adult participation rates ranging from 1% to 9% across regions. Applying these rates to regional populations implies approximately 114,000 active fishers nationally. This estimate is broadly consistent with the 108,000 fishing licences reported by Fish & Game New Zealand (Sanders, 2024).

The average number of trips per active participant was estimated at 15 trips per year. This was derived from reported frequency categories in the Water Safety Survey (weekly, monthly, occasional, once per year). Average trip duration was estimated at 2.4 hours, also based on the 2022 survey.

Monetisation

Non-market values for recreational fishing in New Zealand have been extensively studied (Kaval & Yao, 2007; Marsh & Mkwara, 2013). However, many published estimates are expressed as marginal values (e.g., per fish caught or per incremental change in water quality), which are not directly transferable to a per-trip valuation framework.

A review (G. Kerr, 2003) provides per-trip consumer surplus estimates averaging $40-$70 in 2000 prices. Adjusted to 2025 price levels, this corresponds to approximately $78-$135 per trip. A central estimate of $120 per trip was adopted for modelling purposes, with $80 used as the lower bound.

A newer valuation study (Jiang, 2015) reports consumer surplus estimates for Otago region averaging $605 per domestic trip in 2026 prices. Given the high amenity values associated with Otago's lakes and fisheries, this estimate is likely to be above the national average. This value is therefore adopted as the high estimate.

Trip values were not adjusted for water quality, as water quality effects are already incorporated in the spatial attractiveness surface used to allocate trips.

Data

- Rivermaps water quality and fish presence/absence (https://shiny.niwa.co.nz/nzrivermaps/)
- NZLUM v0.3 (https://lris.scinfo.org.nz/layer/122805-new-zealand-land-use-management-version-03-nzlum-v03/)
- River Environment Classification (REC2) (https://data-niwa.opendata.arcgis.com/maps/3a4b6cc2c1c74fbb8ddbe25df28e410c)
- Predicted lake water quality 2009-13 (https://data.mfe.govt.nz/layer/95541-predicted-lake-water-quality-2009-13/)
- Lake submerged plant index (https://data.mfe.govt.nz/table/104564-lake-submerged-plant-index-1991-2019/)

Steps

- Rasterise conserved land from NZLUM primary code =1
- Run the Python script *recreation_fishing_stream_attractiveness.py* to create stream fishing attractiveness
- Spatially match each LakeSPI data point to a feature in predicted_lake_water_quality
- Rasterise LakeSPI as attractiveness_lake_fishing
- Merge the stream and lake attractiveness layers, taking the maximum if they overlap.
- Run *recreation_calculate_flow.py*

Other freshwater recreation (excluding fishing)

Freshwater ecosystems contribute to recreation by providing the biophysical conditions that enable swimming, paddling and other water-contact activities, including suitable water quality, visual clarity and overall amenity.

Attractiveness

Many studies have shown that water quality indicators such as microbial contamination, clarity, nutrients, and other physico-chemical measures influence participation in a range of water-based recreational activities (Breen et al., 2018). In addition, many freshwater recreation trips involve multiple activities (for example swimming combined with picnicking, paddling, or informal water play) (Galloway, 2012). Therefore, the activity with the highest reported participation rate (swimming/wading) is used as a proxy for general freshwater recreation participation. This layer excludes fishing as it is modelled separately.

Lake and river swimmability categories were converted to relative attractiveness scores (0, 0.30, 0.70, 1.0 for Red/Orange, Yellow, Green, and Blue respectively) using a non-linear scale to reflect the behavioural step-change between moderate-risk and low-risk swimming conditions.

Attractiveness is moderated by accessibility by down-weighting supply cells based on distance to the nearest road. When enabled, attractiveness declines smoothly beyond a defined access threshold (e.g. ~300 m), reflecting that sites further from road access are less likely to be used before demand is allocated via the gravity model.

Demand

The 2022 Water Safety Survey reports that 24% of adults participated in river swimming or wading in the past year. Of the total adult population, 8% reported swimming at least monthly (approximately 12 trips per year), while a further 16% reported participating less frequently. The regional breakdown does not suggest there is much variation between areas (Auckland, Wellington, Canterbury, other North, other South). DOC activity participation data are similar, reporting a 28% annual participation rate for river or lake swimming, although frequency of participation is not reported.

Assuming the occasional group averages four trips per year, this implies an average of 6.7 trips per participant annually and approximately 1.6 trips per capita per year across the adult population ($0.08 \times 12 + 0.16 \times 4 = 1.6$).

A median travel distance of 20 km was assumed, with a kernel cutoff at 65 km, reflecting the predominantly local and often spontaneous nature of river and lake recreation. This is shorter than the distances assumed for mountain biking and day tramping because they are more destination-oriented activities for which participants are typically willing to travel further.

A conservative central value of NZ$116 per trip (2026 NZD) was adopted for general freshwater recreation excluding fishing. Kaval and Yao (2007) report an average non-market benefit of NZ$95 per person per day (2007 NZD) for freshwater-based recreation in New Zealand, but that included some high-value angling. They reported NZ$71 per person per day across all outdoor recreation activities.

Data:

- MfE Lake Water Quality for Swimming Categories, September 2017
- LCDBv6

Marine boating

The ecosystem service provided by marine boating recreation is a cultural service derived from access to navigable coastal and marine environments, generating leisure, enjoyment, and social benefits; however, because most boating trips involve fishing, the associated value also incorporates a provisioning component linked to the opportunity to harvest marine resources, alongside the broader experiential and cultural benefits of being on the water.

Demand

17.6% of adults identify as recreational marine fishers (Heinemann & Gray, 2024) . The Water Safety Survey (2022) similarly reports that 18% of adults participated in coastal boating in the past 12 months, with most boating activity associated with fishing. 18% is therefore adopted for the participation rate.

Trip frequency is estimated using national boating monitors. The Water Safety frequency distributions imply an average of approximately 13 trips per participant per year, while Maritime NZ (2024) implies approximately 11 trips per participant. A midpoint of 12 trips per participant is adopted.

Applying this to the estimated boating population (approximately 0.73 million adults) implies roughly 8.7 million person-trips per year.

Trips are spatially distributed across regions according to Maritime NZ (Ipsos & Maritime New Zealand, 2024) regional boating volumes. Within regions, trips are allocated to SA1 areas based on population.

Flow

Travel behaviour is modelled using a gravity-style distance decay function. The median travel distance is estimated at 35 km based on Water Safety (2022) travel bands. A cutoff of 100 km is applied to capture both typical day-trip behaviour and occasional longer “destination” trips, consistent with approximately 3× the median distance.

Boat ramps are used as spatial accessibility anchors along the coastline. Demand is distributed from population centroids to coastal cells weighted by ramp presence and distance decay.

Within areas covered by MPI’s recreational fishing aerial survey, modelled trip densities are re-weighted to match observed vessel densities. This ensures empirical calibration where observed effort data are available, while preserving the national total trip constraint.

Monetisation

Per-trip recreational values are drawn from New Zealand non-market valuation studies summarised in Kerr and Latham (2011). Reported values, updated to 2026 dollars, range from approximately \$203 to \$339 per trip. These are used as the lower and upper bounds of the valuation range. A central estimate of \$270 per trip is adopted. Total recreational value per hectare is calculated as: annual person-trips × value per trip.

Future improvements

Future improvements could use something similar to the Recreation Opportunity Spectrum (ROS) approach (Richards et al., 2023) to provide a more comprehensive representation of recreation potential. This approach combines characteristics such as landscape attractiveness, proximity to tracks, huts, visitor facilities, protected areas, and other recreation infrastructure to represent the quality and accessibility of recreational opportunities. Recreation scenic quality could also be enhanced through viewshed modelling, particularly in landscapes where panoramic views are an important component of the recreational experience.

7.2. Recreational hunting

Explanation

CICES recognises both wild food provision and outdoor recreation as ecosystem services relevant to hunting. In this assessment, hunting is treated as a recreational service because most of its estimated economic value derives from the hunting experience rather than the harvested meat alone. Any value associated with meat harvest is implicitly included within hunters' consumer surplus, alongside benefits from access to nature, challenge, social connection, cultural practice, and time spent in semi-natural environments. The indicator therefore represents ecosystems' contribution to recreational hunting opportunities supported by free-ranging game populations, rather than a separate estimate of wild meat provision.

Supply indicator

The potential supply indicator represents the ecological capacity of landscapes to sustain harvestable populations of game species and associated recreational hunting opportunities. Potential supply was modelled using the Department of Conservation species extent layers for the six major big-game species included in Kerr and Abell (2014): red deer, sika deer, pigs, goats, tahr, and chamois. These layers provide mapped species presence and relative abundance categories. Other hunted species (such as possums, rabbits, wallabies, and goats) were excluded because they are more often associated with pest control than recreational hunting value.

Relative supply weights were assigned to the DOC abundance classes to create a quantitative indicator of provisioning potential. Cells classified as High abundance were assigned a value of 100, Medium abundance 60, Low abundance 25, and cells with recorded presence but null abundance 10. Areas outside mapped species extent were assigned zero supply. These values represent relative hunting opportunity rather than actual animal density.

The DOC extent layers implicitly capture the broad habitat associations of each species. Deer and pigs are primarily associated with forest, scrub, regenerating vegetation, and forest-grassland mosaics. Goats are most common in shrubland, dry hill country, and rugged scrub-grass environments. Tahr and chamois are strongly associated with alpine and subalpine environments in the South Island high country.

New Zealand game birds are predominantly freshwater-associated species such as ducks, shelduck, swan, and pūkeko. Highest provisioning capacity occurs in wetlands, lakes, rivers, estuarine margins, and agricultural landscapes with ponds and riparian habitat (McDougall et al., 2018). Open water, wet pasture, and adjacent feeding areas are critical. Marine environments do not support game bird species in New Zealand, although some freshwater species utilise estuarine zones. The game bird hunting supply raster was operationalised as LCDB classes for lake & pond, river, estuarine, herbaceous freshwater and herbaceous saline. Urban areas (as defined by Statistics New Zealand IFUA classes 101,102, and 103) were excluded from hunting supply).

Flow indicator

The realised flow indicator represents the actual annual harvest of wild game and the associated recreational values. Hunting value is calculated separately for seven species because species occupy different environments and attract different consumer surplus values.

Regional hunting participation rates were sourced from the ActiveNZ survey (Sport New Zealand, 2025). The resulting total participation estimate of 45,000 adults is consistent with Kerr and Abell's (2014) earlier estimate of 30,000 to 50,000 big-game hunters in New Zealand.

Species shares of trips and trips per hunter per year were sourced from Kerr and Abell (2014). Because many hunters target multiple species, species-specific participation rates sum to more than 100%. To avoid double counting when allocating total recreational hunting value across species, total trips were proportionally normalised by dividing by the sum of species shares, so that aggregate species-attributed trips summed to total hunting activity. Later species-specific valuation studies reported slightly different shares and distances for sika deer (G. N. Kerr & Abell, 2016) and Himalayan tahr (G. Kerr, 2019) hunting; however, the values from Kerr and Abell (2014) were retained to maintain methodological consistency across all modelled species. The differences are minor relative to the broader uncertainty surrounding the monetary valuation estimates.

Gamebird hunters were treated separately. The estimated total of 35,000 gamebird hunters was informed by Fish & Game New Zealand licence statistics (Fish & Game New Zealand, 2023). These 35,000 were distributed across regions as per the region of residence in figure 1 of the same report. Although some gamebird hunters also hunt big game (Nugent, 1992), these activities are generally undertaken on separate trips. Because values are trip-based, there is unlikely to be the same double-counting issue across big-game and gamebird hunting as there is among big-game species.

Trips per year by species were sourced from Kerr and Abell (2014) for big game and Fish & Game New Zealand (2023) for gamebirds. Table 17 presents the estimated total trips for each species, calculated as population multiplied by species share and trips per year, divided by the sum of species shares.

Table 17: Hunting population, trips, and average distance travelled.

Species	Hunter population	Species share of trips	Trips per hunter/year	Days per hunt	Avg distance one-way (km)	Trips per year
Red deer	45,000	64%	15.9	2.2	121.0	303,630
Sika deer	45,000	11%	15.8	2.5	156.0	51,890
Fallow deer	45,000	17%	16.8	2.0	123.8	85,435
Pigs	45,000	37%	18.7	1.9	91.0	207,099
Tahr	45,000	7%	17.2	3.5	270.1	36,122
Chamois	45,000	11%	17.5	2.9	200.2	57,487
Game birds	35,000	N/A	15.4	1.0	50.0	540,050

Hunting effort was spatially allocated by convolving the SA1-level hunter population with an exponential distance-decay kernel across suitable hunting locations, informed by the DOC species extent layers. Big-game hunting is assumed to occur only within DOC permit areas. Bird hunting is not restricted to DOC permit areas, but is restricted to relevant land-cover classes that are typically publicly accessible (lake, river, estuarine, herbaceous freshwater, and herbaceous saltwater), acknowledging that this excludes hunting on private farm ponds and other minor habitats.

Monetisation

The valuation component represents the recreational consumer surplus generated by hunting opportunities. Values are based on the recreational hunting experience rather than the market value of meat harvested. This reflects evidence that the primary value derived from hunting in New Zealand is recreational and experiential, including challenge, access to nature, social participation, and cultural identity, rather than food provisioning alone (G. Kerr & Abell, 2014).

Where available, New Zealand-specific valuation studies are used. For species without direct New Zealand estimates, values are drawn from the international benefit-transfer and meta-analytic valuation function estimated by Kerr and Woods (2010). All values were expressed as consumer surplus per hunting trip and converted to 2026 NZD.

Red deer, fallow deer, and chamois were assigned the generic ungulate hunting benefit-transfer value estimated by Kerr and Woods (2010). This estimate was derived using a meta-analytic transfer approach based on international hunting valuation studies. Corresponding low and high values were taken from the uncertainty range reported in the same study.

Sika deer values were sourced from the species-specific choice modelling study by Kerr and Abell (2016). That study identified substantial variation among hunter groups, with local hunters exhibiting lower willingness to pay than enthusiast hunters. The low and high estimates used here correspond to the range implied by those latent classes.

Himalayan tahr values were sourced from Kerr (2019). As with the sika study, low and high estimates reflect the variation observed across different hunter classes.

No New Zealand-specific consumer surplus estimates were found for pig hunting. Pig hunting values were therefore anchored at approximately 50% of the deer hunting value based on international evidence indicating that wild pig and wild boar hunting tends to generate lower willingness to pay than deer hunting (Engelmann et al., 2016). This likely reflects the more localised, less trophy-oriented nature of pig hunting relative to large ungulate hunting.

Gamebird hunting values were estimated using the meta-regression coefficients reported by Gren and Kerr (2022). Their model estimated that waterfowl hunting days generate consumer surplus values approximately 37% lower than deer hunting days. A further adjustment was applied to account for trip duration, because gamebird hunting in New Zealand is typically undertaken as single-day outings (Nugent, 1992), whereas deer hunting trips average 2.2 days.

Table 18 presents the resulting consumer surplus values per hunting trip in both original study-year prices and inflation-adjusted 2026 NZD values.

Table 18: Hunting trip consumer surplus values in original prices and 2026 prices

	Value per hunt (original prices)			Value per hunt (NZD 2026)		
Species	**Central**	**Low**	**High**	**Central**	**Low**	**High**
Red deer	$487	409	565	$682	$573	$791
Sika deer	$993	700	1300	$1,390	$980	$1,820
Fallow deer	$487	409	565	$682	$573	$791
Tahr	$827	$230	$1,347	$1,158	$322	$1,886
Pigs	$244	$205	$283	$341	$286	$396
Chamois	$487	409	565	$682	$573	$791
Game birds	$84	$70	$97	$117	$98	$136

The final step allocates trips generated by hunter populations to suitable supply cells for each species. SA1-level hunter populations are convolved with an exponential distance-decay kernel, so that hunting locations closer to the population source receive a larger share of expected trips. As noted above, some big-game trips may target multiple species; this is addressed through the species-level trip normalisation applied in the preceding step. Species-specific trip values are then summed to produce the final realised flow value raster.

Limitations

The main limitations are data age and spatial precision. The DOC species extent layers and several hunting participation surveys are relatively old, so current species distributions, abundance, access, and hunting behaviour may differ from the patterns represented here. Some valuation parameters also rely on benefit transfer rather than New Zealand species-specific estimates.

The spatial allocation indicates where hunting effort is likely to occur, but does not capture other factors that influence the desirability of particular hunting areas, such as terrain, access quality, crowding, trophy potential, landowner relationships, or local knowledge. The model also allocates all big-game hunting effort to DOC land, which omits hunting on private land and may understate the importance of some lowland and farm-based hunting areas. The resulting rasters should therefore be interpreted as broad indicators of recreational hunting value, not precise estimates of current hunting pressure.

Data

- DOC Recreational Hunting Permit Areas
- DOC Red_Deer_Distribution_2014
- DOC Sika_Deer_Distribution_2007
- DOC Feral_Pig_Distribution_2007
- DOC Fallow_Deer_Distribution_2007
- DOC Himalayan_Tahr_Distribution_2014
- DOC Chamois_Distribution_2007
- lris-lcdb-v60-land-cover-database-version-60-mainland-new-zealand
- functional-urban-area-2023-generalised

Steps

- Select functional urban area features with IFUA class 101,102,103 and delete holes so that urban lakes are counted as urban. Rasterise the inverse as *rural_land.tif*
- Rasterize LCDB6 classes 20,21,22,45,46 (water and wetlands) with a burn value of 1 and save as *bird_habitat.tif*
- Multiply bird_habitat*rural_land and save as *hunting_game_bird_supply.tif.*
- Rasterize DOC hunting permit areas with a burn value of 1 and save as *hunting_permit_areas.tif*
- Create the file *region_participation.csv* with the fields REGC2020, participation_big_game and participation_bird_game
- Create the file *total_value.csv* with fields for species, trips, mean distance, and total value central low and high.
- Run the python file *Hunting_allocate_value.py*

7.3. Aesthetics

Explanation

Landscape aesthetics is the ecosystem service by which natural and semi-natural environments provide visual amenity, scenic beauty, and a sense of place to people who live, work, or travel nearby. Unlike active recreation, this service is experienced passively - through windows, during commutes, and in the course of daily life. It has value even when people are not consciously seeking out nature.

New Zealand has a fragmented literature on landscape aesthetic values. Existing studies provide evidence that people value scenic amenity, water views, open space, native vegetation and particular rural landscape features, using methods such as visual preference surveys, hedonic pricing and contingent valuation (G. Brown & Brabyn, 2012; P. Brown & Mortimer, 2014; Fernandez, 2019; Samarasinghe & Sharp, 2008; Swaffield & McWilliam, 2013). However, these studies are generally site-specific, method-specific or focused on particular amenities, and there no comprehensive assessment of landscape monetary values for different ecosystems. The layer developed here should therefore be treated as a rapid indicative assessment: it combines available scenic-quality proxies, exposure modelling and transferred valuation evidence to produce a first-order national estimate.

Supply

The supply indicator represents the scenic experience a person would have if they stood in each cell, expressed as a dimensionless index from 0 to 100. It is constructed from four components, consistent with established approaches to landscape aesthetic assessment (Frank et al., 2013) and the spatial representation of cultural ecosystem services in New Zealand (G. Brown & Brabyn, 2012; Swaffield & McWilliam, 2013).

Structural scenic quality is derived from a landcover base score modified by tree canopy cover, reflecting New Zealand visual-preference evidence that trees and shelterbelts increase perceived landscape quality, while more intensive agricultural features can reduce it (P. Brown & Mortimer, 2014).

Topographic drama is represented by slope, with steeper terrain assigned higher scenic scores up to a reference angle of 30 degrees. Both structural quality and topographic drama are convolved with a Gaussian spatial kernel (sigma = 300 m, truncated at 1 km) before being combined, because a viewer's visual experience is determined by the surrounding landscape rather than only the cell they occupy. This neighbourhood-based representation reflects the understanding that scenic and aesthetic values are emergent properties of landscape context rather than single pixels (Swaffield, 2012).

Landscape complexity is captured by a Shannon diversity index and patch density score, both calculated at a 3 km radius and therefore already neighbourhood-representative, so no further spatial smoothing is applied.

Water proximity is represented by an exponential distance-decay function centred on each cell, reflecting empirical evidence that proximity to rivers, lakes, and the coast enhances perceived landscape value and influences spatial patterns of social landscape values (Samarasinghe & Sharp, 2008).

The four components are combined as a weighted sum with weights of 0.40, 0.20, 0.25, and 0.15 for structure, topography, complexity, and water proximity respectively, consistent with integrative landscape aesthetic assessment frameworks (Frank et al., 2013; Swaffield, 2012).

Monetisation

The realised benefit of aesthetic value is calculated as the product of exposure hours, experienced scenic quality, and a water proximity uplift applied at the viewer's location. Exposure hours are derived from resident adult population, assuming 8 hours per day, and state highway traffic, assuming 6 seconds per 100 m cell at 1.4 persons per vehicle.

A value of NZ$1.37 per person-hour is applied to exposure to landscapes of the highest scenic quality. For landscapes with lower scenic quality, this value is scaled proportionally using the scenic quality score (0-100). The value is intended to represent the welfare benefit associated with experiencing attractive landscapes rather than recreation or tourism activity.

The valuation is based on Brander and Koetse (2011), who synthesised contingent valuation studies of urban and peri-urban open space and reported an average value of approximately NZ$4,000 per hectare per year (converted from the original study values). This annual value was converted to an hourly exposure value by assuming 2,920 exposure hours per adult per year (8 hours per day). Although some New Zealand studies have examined aesthetic values in specific settings, such as coastal views, no broadly transferable New Zealand estimates were identified, so international benefit-transfer evidence was used.

Sensitivity values are also derived from Brander and Koetse (2011). The low value of NZ$0.99 per person-hour is based on the median value reported in their dataset, while the high value of NZ$11.68 per person-hour is based on the mean value. The wide range reflects substantial variation in aesthetic valuation estimates across studies and contexts.

Limitations

The current method estimates aesthetic benefits based on scenic quality within a fixed distance of where people live or travel. As a result, large and visually prominent landscape features such as mountains, volcanic cones, or distant coastlines may be underrepresented, even where they strongly influence perceived landscape character over long viewing distances. The model also uses simplified proximity and neighbourhood-based assumptions rather than explicitly modelling line-of-sight visibility, atmospheric conditions, viewing angles, or obstructions. In addition, the valuation component relies on transferred evidence from a relatively limited New Zealand literature on landscape amenity and environmental aesthetics, as no established national spatial layer of aesthetic ecosystem service values currently exists. The resulting estimates should therefore be interpreted as indicative relative values.

Future improvements

Future work could improve the representation of aesthetic ecosystem services through explicit viewshed modelling using high-resolution elevation and vegetation data, allowing distant but visually dominant landforms to contribute appropriately to scenic experience. The valuation component could also be strengthened through New Zealand-specific stated preference or hedonic pricing studies designed explicitly for landscape aesthetics, enabling more robust estimation of monetary values and uncertainty ranges.

7.4. Terrestrial biodiversity

Explanation

This service is interpreted as the value derived from the continued existence of indigenous land cover and the ecological landscapes that underpin historical, cultural, and identity-based connections to place. In the absence of spatially explicit data on cultural sites, practices, or ecosystem condition, indigenous land cover is used as a proxy for the biophysical foundation of cultural heritage value. This approach recognises that native ecosystems contribute to sense of place, historical continuity, and intergenerational identity, but it does not capture variation in ecosystem state or the condition of culturally significant sites. As a result, the indicator reflects the presence of culturally meaningful ecological landscapes rather than their current quality.

Unlike the recreation layers, which vary spatially according to observed participation and travel behaviour, the existence value layer applies an average non-use value across all New Zealand adults, reflecting the assumption that biodiversity existence values are broadly held by the national population regardless of whether individuals directly visit or experience the ecosystem.

Existence values were assigned only to wetlands and native forest because these ecosystems are both highly threatened and strongly represented in non-market valuation literature.

Wetlands

Wetland biodiversity existence values are highly uncertain and strongly context-specific, so the layer uses a simple sensitivity-based benefit transfer approach rather than a single point estimate. All source values were converted to NZD 2026. The low value is based on Patterson and Cole's passive wetland value of NZ$1,667 per hectare, which itself was derived from two United States studies. The high value uses Kirkland's contingent valuation study of Whangamarino wetland, the only New Zealand wetland CV study identified, at approximately NZ$16,000 per hectare. The central estimate uses de Groot et al.'s global synthesis value for wetland biodiversity of NZ$2,814 per hectare.

Other evidence was reviewed but not used directly: a global meta-analysis (Subroy et al., 2019) reports biodiversity values on a per-species basis, which is not suitable for aggregation to a per-hectare layer. The New Zealand literature on willingness to pay for biodiversity improvements, summarised by Tait et al. (2017), generally relates to specific policy outcomes rather than total per-hectare existence values.

The resulting per-hectare values are scaled by wetland scarcity, measured as the percentage of wetland remaining. Areas where less than 1% of the original wetland extent remains receive the full value, while areas where 50% or more remains receive a 50% lower value, with intermediate values scaled between these bounds. The wetland values are applied to LCDBv6 features classed as wetland according to the attribute wetland_23.

Indigenous forest

Indigenous is defined as LCDBv6 classes “indigenous forest”, ”Broadleaved Indig”, and “Indigenous Forest”. The central value used is NZ$2,077/ha/year, taken from de Groot’s (2012) genetic diversity value for temperate forest. The low value is NZ$346.86/ha/year, based on Patterson and Cole 2013 passive value for forest parks, and the high value is NZ$4,128.48/ha/year, based on Patterson and Cole’s passive value for national parks.

These base values are scaled by the percentage of natural cover remaining. The multiplier is 1.0 where less than 1% remains, 0.95 where less than 5% remains, 0.85 where less than 10% remains, 0.75 where less than 25% remains, 0.6 where less than 50% remains, and 0.5 otherwise. Wetlands take priority where forest and wetland overlap.

Data

- Iris-threatened-environments-classification-2012
- Iris-lcdb-v60-land-cover-database-version-60-mainland-new-zealand
- Prediction of wetlands before humans arrived (https://data.mfe.govt.nz/layer/52677-prediction-of-wetlands-before-humans-arrived/)
- eco-index-catchments-for-nz (https://eco-index.koordinates.com/layer/115002-eco-index-catchments-for-nz/)

Steps

- Select indigenous LCDB classes and rasterise with a burn value of 1
- Rasterise Nat12Pct from threatened environments
- Rasterise LCDBv6 wetland_23 features as current wetland
- Rasterise prehuman wetlands
- For each eco-index catchment, calculate the area of current wetland/prehuman wetland and rasterise as wetland_remaining_pct
- Run the Python script *natural_diversity_land_calculate_flow.py* to allocate and scale the values.

Limitations

Potential biodiversity datasets and tools were identified during the project, including species-prioritisation information associated with DOC programmes. However, these data were not available in a form that could be accessed or incorporated within the project timeframe. As a result, the biodiversity component relies on a simplified spatial proxy, treating native forest area as the main indicator of biodiversity value. This is likely to understate spatial variation in biodiversity significance, particularly where some areas provide habitat for threatened or priority species.

A further limitation is that existence values are currently assigned only to wetlands and native forest, and therefore do not capture biodiversity and natural heritage values associated with other indigenous ecosystems such as shrublands, tussock grasslands, dunes, braided rivers, geothermal systems, and alpine environments.

7.5. Marine biodiversity

Explanation

Marine natural diversity represents the ecosystem service associated with the existence and maintenance of biodiversity in marine ecosystems. In ecosystem service terms, this primarily reflects non-use values such as existence and bequest values, where people derive benefit simply from knowing marine species and ecosystems persist. Because this value does not depend on direct interaction with the ecosystem, supply and realised flow are effectively the same.

Marine natural diversity is assessed separately from land-based biodiversity because marine ecosystems differ substantially in their ecological structure and cultural drivers of value.

The supply of marine natural diversity is estimated using the Department of Conservation's Seafloor Community Classification (SCC). The geographic extent is the Exclusive Economic Zone (EEZ) of New Zealand. The SCC groups areas of the seafloor according to environmental characteristics and provides estimates of the expected composition of biological communities associated with each class. Mean taxa occurrence values are available for species groups: macroalgae, demersal fish, benthic invertebrates, and reef fish. These data are used to characterise the biodiversity associated with each seafloor class.

Biodiversity was summarised using the Shannon diversity index, a commonly used diversity metric that incorporates both taxa richness and relative abundance. Diversity metrics of this type are widely applied as indicators of ecosystem condition in ecosystem accounting frameworks (United Nations, 2022). The index is calculated separately for each species group and then combined to produce an overall biodiversity score for each seafloor class.

Marine reserves contribute to the protection of biodiversity and support the functioning of marine ecosystems by reducing extractive pressures. However, the indicator used here measures the observed or expected current state of biodiversity across marine habitats rather than the effectiveness of specific protection areas. Because detailed taxa measurements are not consistently available within individual marine reserves, biodiversity values are allocated across all seafloor community classes using the SCC dataset.

Monetisation

Estimates of marine non-use values are available internationally, but these are not directly transferable to the New Zealand context. Australian studies frequently focus on tropical coral reef systems such as the Great Barrier Reef, where biodiversity is strongly linked to tourism and recreation values that are not representative of most New Zealand marine ecosystems. Similarly, UK valuation studies reflect social preferences and marine environments that differ substantially from those in New Zealand. As a result, transferring these values would likely misrepresent the relative importance of marine biodiversity to New Zealand households.

Although the New Zealand literature on marine non-use values is limited, two studies provide relevant estimates. Chhun et al. (2013) conducted a choice experiment examining preferences for management of customary marine areas and included an attribute representing biodiversity condition. The estimated willingness to pay for a "medium" level of biodiversity improvement was approximately $22.33 per household per year in 2026 prices (standard deviation $2–$31), while a "good" biodiversity outcome was valued at around $42 per household (range $7.25–$54). Rojas-Nazar (2022) used a contingent valuation approach to estimate willingness to pay for marine reserves in New Zealand and reported values between approximately $18.57 and $24.75 per household per year in 2025 prices, which is broadly consistent with the earlier study.

Based on the consistency between these estimates, this analysis adopts $22 per household per year as the central estimate of the non-use value of marine biodiversity. A low-high range of $7-$54 per household per year is used to represent uncertainty around this estimate. Because the underlying studies value biodiversity protection in specific marine areas, particularly marine reserves, the resulting national value is allocated across the entire marine environment according to the calculated biodiversity score for each seafloor community classification area. This approach may underestimate total non-use values if households place additional value on the protection of biodiversity across a broader range of marine ecosystems. However, stated preference studies often find limited sensitivity of willingness-to-pay estimates to the size of the area protected, suggesting that values may be driven more by the existence of biodiversity protection than by the precise extent of the protected area.

Data

- DOC Seafloor Community Classification and associated mean taxa data

Steps

- Use the combined (Exclusive Economic Zone + Territorial Sea) and reprojected SCC raster created for the seafood provisioning indicator
- Save the mean taxa data in a csv with columns for group, species, and each SCC class
- Run the Python script *Cultural_calculate_marine_biodiversity_score.py* to calculate the scores and allocate values.

8. Limitations

The ecosystem service layers developed for this project represent a pragmatic national-scale assessment based on available data, time, and resource constraints. While considerable effort has been made to ensure consistency, transparency, and policy relevance, the results should be interpreted in light of several important limitations.

8.1. Data limitations

The assessment relies primarily on nationally consistent datasets to enable complete spatial coverage and comparability across New Zealand. In many cases, higher-resolution or more current local datasets exist that could improve the representation of ecosystem condition, service supply, or beneficiary populations. Due to project scope constraints, these local datasets were generally not incorporated.

Input datasets also vary in age, quality, completeness, and spatial resolution. Some indicators are based on direct observations, while others rely on modelled or derived datasets. Consequently, uncertainty in input data propagates through to the final ecosystem service estimates.

Data coverage is also variable for outer islands, including the Chatham Islands. Where one or more input layers have missing values, these can propagate through the modelling process and result in missing or zero values in the final ecosystem service layers. An absence of mapped value on outer islands should not be interpreted as evidence that ecosystem services are absent.

8.2. Modelling simplifications

Ecosystem service provision arises from complex ecological, hydrological, social, and economic processes that vary across space and time. National-scale modelling necessarily simplifies many of these processes. In some cases, simplified proxy indicators were used where more detailed process-based modelling was not feasible. As a result, the layers should be viewed as indicative representations of ecosystem service patterns rather than precise predictions of service delivery.

The assessment generally represents current conditions and does not explicitly model dynamic ecological responses, threshold effects, lagged responses, climate change impacts, or future land-use changes unless these are incorporated within specific indicators.

Some services are likely to be underrepresented because the available biophysical data do not capture all relevant ecological pathways. For example, marine reserves may contribute to commercial fisheries through spillover effects and broader ecosystem functions, but these contributions are not explicitly recognised in the current layers where suitable national-scale biophysical evidence is unavailable.

8.3. Spatial resolution and area representation

All layers have been harmonised to a common spatial framework to support aggregation and comparison. However, the nominal grid resolution should not be interpreted as implying equivalent precision at that scale. Many underlying datasets are a lot coarser than the final output resolution and the cells can represent average conditions over larger areas.

The layers are generally interpreted on a planimetric area basis. This means that hilly or mountainous areas are represented by their horizontal mapped area rather than their true three-dimensional surface area. In reality, sloping land has a larger surface area than flat land within the same mapped footprint. This may lead to some underrepresentation of services that depend directly on surface area, such as vegetation cover, habitat extent, soil processes, or above-ground biomass.

The layers are therefore most appropriate for regional, catchment, or national-scale analyses and should not be used as the sole basis for site-level decision-making, resource consent processes, engineering design, or detailed ecological assessment.

8.4. Ecosystem service coverage

The assessment does not represent all ecosystem services. Some services were excluded because suitable national datasets, modelling approaches, or valuation evidence were unavailable. Others are only partially represented due to limitations in current scientific understanding or data availability.

Cultural ecosystem services are particularly challenging to quantify spatially because they often depend on personal, cultural, historical, and relational values that cannot be fully captured through national datasets. Similarly, some biodiversity, existence, option, and bequest values remain difficult to represent spatially using currently available methods.

8.5. Valuation limitations

Monetary valuation estimates are derived using a range of approaches, including market prices, avoided costs, replacement costs, production-based methods, and benefit transfer. These methods involve assumptions that may not hold equally across all locations and contexts.

Many values should be interpreted as approximate indicators of the welfare implications of marginal changes in ecosystem service provision rather than precise estimates of economic value. The values are not intended to represent the total value of ecosystems, nature, biodiversity, or natural capital assets.

Where benefit transfer methods have been used, transferred values may not fully reflect local ecological, social, cultural, or economic conditions. Although values have been adjusted where possible, uncertainty remains regarding their applicability to specific locations or policy contexts.

8.6. Double counting and service interactions

The framework focuses on final ecosystem services to minimise double counting. However, ecosystem services are inherently interconnected, and complete separation of ecological processes and benefits is not always possible. Some residual overlap between indicators may remain despite efforts to define service boundaries consistently.

In addition, ecosystem services may interact through synergies, trade-offs, and non-linear relationships that are not fully represented within individual layers.

8.7. Uncertainty

Uncertainty is present throughout the assessment and arises from data limitations, modelling assumptions, ecological variability, valuation evidence, and future conditions. The uncertainty associated with different layers varies substantially and cannot always be quantified formally.

For several indicators, low and high sensitivity estimates have been provided to illustrate the plausible range of results. However, these ranges should not be interpreted as formal confidence intervals. They are intended to provide an indication of the sensitivity of results to key assumptions rather than a complete characterisation of uncertainty.

8.8. Appropriate interpretation

The ecosystem service layers are best viewed as decision-support tools designed to improve the visibility of environmental benefits and costs within policy analysis, cost-benefit analysis, spatial planning, and strategic assessment. They are most appropriate for comparing options, identifying broad spatial patterns, and informing discussion of environmental trade-offs. They should be used alongside ecological expertise, local knowledge, mātauranga Māori, stakeholder engagement, and other relevant evidence rather than as a substitute for them.

9. References

Alfaro, A., Jeffs, A., Gardner, J., Bollard Breen, B., & Wilkin, J. (2011). Green-lipped mussels in GLM 9. *New Zealand Fisheries Assessment Report*, *48*, 80.

American Society of Agricultural Engineers. (2005). *Manure Production and Characteristics* (ASAE D384.2 MAR2005). ASAE-The Society for engineering in agricultural, food, and biological systems, Joseph, MI, USA. https://www.researchgate.net/profile/Rb-Brobst/post/How-much-manure-does-1-pig-produce/attachment/5c4f2679cfe4a781a57aa56a/AS%3A720076667904003%401548691064961/download/ASAE_D384.2.pdf

Apiculture New Zealand. (2024). *New Zealand Honey Strategy 2024–2030*. Apiculture New Zealand. https://apinz.org.nz/wp-content/uploads/2024/02/NZ-Honey-Strategy-2024-2030-FINAL-version-for-web-20-Feb.pdf

Aqualinc. (2021). *WATER AVAILABILITY A ND SECURITY : National-Scale Assessment* (Prepared for Ministry for Primary Industries MPI Technical Paper No: 2021/17).

Astell-Burt, T., Feng, X., & Kolt, G. S. (2014). Green space is associated with walking and moderate-to-vigorous physical activity (MVPA) in middle-to-older-aged adults: Findings from 203 883 Australians in the 45 and Up Study. *British Journal of Sports Medicine*, *48*(5), 404–406. https://doi.org/10.1136/bjsports-2012-092006

Ausseil, A. G. E., Dymond, J. R., Kirschbaum, M. U. F., Andrew, R. M., & Parfitt, R. L. (2013). Assessment of multiple ecosystem services in New Zealand at the catchment scale. *Environmental Modelling & Software*, *43*, 37–48. https://doi.org/10.1016/j.envsoft.2013.01.006

Bagstad, K. J., Johnson, G. W., Voigt, B., & Villa, F. (2013). Spatial dynamics of ecosystem service flows: A comprehensive approach to quantifying actual services. *Ecosystem Services*, *4*, 117–125. https://doi.org/10.1016/j.ecoser.2012.07.012

Barbier, E. B., Georgiou, I. Y., Enchelmeyer, B., & Reed, D. J. (2013). The value of wetlands in protecting southeast Louisiana from hurricane storm surges. *PloS One*, *8*(3), e58715.

Berthelsen, A., Bulmer, R., Stewart-Sinclair, P., & Lundquist, C. (2025). *Developing a national dataset for coastal wetland blue carbon in Aotearoa New Zealand* (Cawthron Report 4161A).

Blackwell, B. (2007). The Value of a Recreational Beach Visit: An Application to Mooloolaba Beach and Comparisons With Other Outdoor Recreation Sites**This paper does not necessarily reflect the views of the NCME or AMC. *Economic Analysis and Policy*, *37*(1), 77–98. https://doi.org/10.1016/S0313-5926(07)50005-6

Brander, L. M., & Koetse, M. J. (2011). The value of urban open space: Meta-analyses of contingent valuation and hedonic pricing results. *Journal of Environmental Management*, *92*(10), 2763–2773. https://doi.org/10.1016/j.jenvman.2011.06.019

Breen, B., Curtis, J., & Hynes, S. (2018). Water quality and recreational use of public waterways. *Journal of Environmental Economics and Policy*, *7*(1), 1–15. https://doi.org/10.1080/21606544.2017.1335241

Broussard, M. A., & Pattemore, D. E. (2023). Kiwifruit Pollination. In A. Richardson, J. Burdon, & R. Ferguson (Eds.), *Botany, Production and Uses* (pp. 207–227). CABI. https://doi.org/10.1079/9781800620933.0013

Brown, G., & Brabyn, L. (2012). The extrapolation of social landscape values to a national level in New Zealand using landscape character classification. *Applied Geography*, *35*(1–2), 84–94.

Brown, P., & Mortimer, C. (2014). Econometric analysis of landscape preferences in Canterbury, New Zealand. *Economics Research International*, *2014*(1), 259471.

Burkhard, B., Kandziora, M., Hou, Y., & Müller, F. (2014). Ecosystem service potentials, flows and demands-concepts for spatial localisation, indication and quantification. *Landscape Online*, *34*, 1–32. https://doi.org/10.3097/LO.201434

Burkhard, B., & Maes, J. (Eds.). (2017). *Mapping Ecosystem Services*. Pensoft Publishers. https://doi.org/10.3897/ab.e12837

Chalkiadakis, C., Drakou, E. G., & Kraak, M.-J. (2022). Ecosystem service flows: A systematic literature review of marine systems. *Ecosystem Services*, *54*, 101412. https://doi.org/10.1016/j.ecoser.2022.101412

Chapman, D. F., Mackay, A. D., Caradus, J. R., Clark, D. A., & Goldson, S. L. (2025). Pasture productivity in New Zealand 1990–2020: Trends, expectations, and key factors. *New Zealand Journal of Agricultural Research*, *68*(6), 1221–1264. https://doi.org/10.1080/00288233.2024.2425071

Chhun, S., Thorsnes, P., & Moller, H. (2013). Preferences for Management of Near-Shore Marine Ecosystems: A Choice Experiment in New Zealand. *Resources*, *2*(3), 406–438. https://doi.org/10.3390/resources2030406

Daigneault, A., Eppink, F. V., & Lee, W. G. (2017). A national riparian restoration programme in New Zealand: Is it value for money? *Journal of Environmental Management*, *187*, 166–177. https://doi.org/10.1016/j.jenvman.2016.11.013

DairyNZ. (2025). *DairyNZ Economic Survey 2023-24*. https://www.dairynz.co.nz/media/knchednv/dairynz-economic-survey-2023-24.pdf

De Groot, R., Brander, L., Van Der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., Hussain, S., Kumar, P., McVittie, A., Portela, R., Rodriguez, L. C., Ten Brink, P., & Van Beukering, P. (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services*, *1*(1), 50–61. https://doi.org/10.1016/j.ecoser.2012.07.005

Denne, T., & Wright, L. (2017). *Economic value of river control, flood protection and drainage (RCFPD) schemes in New Zealand* (Covec report prepared for River Managers' Special Interest Group).

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. https://doi.org/10.1016/j.forpol.2011.11.006

Dominati, E., Mackay, A., Green, S., & Patterson, M. (2014). A soil change-based methodology for the quantification and valuation of ecosystem services from agro-ecosystems: A case study of pastoral agriculture in New Zealand. *Ecological Economics*, *100*(Supplement C), 119–129. https://doi.org/10.1016/j.ecolecon.2014.02.008

Dymond, J. R., Ausseil, A.-G. E., Ekanayake, J. C., & Kirschbaum, M. U. (2012). Tradeoffs between soil, water, and carbon–a national scale analysis from New Zealand. *Journal of Environmental Management*, *95*(1), 124–131.

Dymond, J. R., Betts, H. D., & Schierlitz, C. S. (2010). An erosion model for evaluating regional land-use scenarios. *Environmental Modelling & Software*, *25*(3), 289–298. https://doi.org/10.1016/j.envsoft.2009.09.011

Edmeades, D., & McBride, R. (2025). Is soil fertility a factor limiting pasture production and persistence in New Zealand? *Journal of New Zealand Grasslands*, *87*, 79–85. https://doi.org/10.33584/jnzg.2025.87.3723

Egan, A. L., Rolfe, J., Cassells, S., & Chilvers, B. L. (2023). Potential changes in the recreational use value for Coastal Bay of Plenty, New Zealand due to oil spills: A combined approach of the travel cost and contingent behaviour methods. *Ocean & Coastal Management*, *228*, 106306.

Engelmann, M., Lagerkvist, C.-J., & Gren, I.-M. (2016). *Hunting value of wild boar in Sweden: A choice experiment*. https://swopec.hhs.se/slueko/abs/slueko2016_001.htm

European Commission. Joint Research Centre. (2017). *Global flood depth-damage functions: Methodology and the database with guidelines.* Publications Office. https://doi.org/10.2760/16510

European Commission Joint Research Centre. (2024). *The contribution of ecosystem services in agricultural production: An application of the production function approach.* Publications Office. https://doi.org/10.2760/1432445

Fairchild, T. P., Bennett, W. G., Smith, G., Day, B., Skov, M. W., Möller, I., Beaumont, N., Karunarathna, H., & Griffin, J. N. (2021). Coastal wetlands mitigate storm flooding and associated costs in estuaries. *Environmental Research Letters*, *16*(7), 074034.

Fernandez, M. A. (2019). *A review of applications of hedonic pricing models in the New Zealand housing market*. Auckland, New Zealand. https://knowledgeauckland.org.nz/media/1101/dp2019-002-review-hedonic-pricing-models-nz-housing-market.pdf

Fernandez, M. A., & Daigneault, A. (2017). Erosion mitigation in the Waikato District, New Zealand: Economic implications for agriculture. *Agricultural Economics*, *48*(3), 341–361.

Frank, S., Fürst, C., Koschke, L., Witt, A., & Makeschin, F. (2013). Assessment of landscape aesthetics—Validation of a landscape metrics-based assessment by visual estimation of the scenic beauty. *Ecological Indicators*, *32*, 222–231. https://doi.org/10.1016/j.ecolind.2013.03.026

Freeman Iii, A. M., Herriges, J. A., & Kling, C. L. (2014). *The measurement of environmental and resource values: Theory and methods*. Routledge.

Galloway, S. (2012). Recreation Specialization Among New Zealand River Recreation Users: A Multiactivity Study of Motivation and Site Preference. *Leisure Sciences*, *34*(3), 256–271. https://doi.org/10.1080/01490400.2012.669690

Ghasemi, M., González-García, A., & Serrao-Neumann, S. (2025). Ecosystem services modelling to analyse the isolation of protected areas from a social-ecological perspective. *Journal of Environmental Management*, *386*, 125459. https://doi.org/10.1016/j.jenvman.2025.125459

Goodwin, M. (2012). *Pollination of crops in Australia and New Zealand*. RIRDC Wagga Wagga, Australia. https://www.academia.edu/download/58636745/12-059.pdf

Gren, I.-M., & Kerr, G. (2022). A Meta-Regression Analysis of Hunters' Valuations of Recreational Hunting. *Sustainability*, *15*(1), 27. https://doi.org/10.3390/su15010027

Hamel, P., & Bryant, B. P. (2017). Uncertainty assessment in ecosystem services analyses: Seven challenges and practical responses. *Ecosystem Services*, *24*, 1–15.

Hanemann, W. M. (1994). Valuing the environment through contingent valuation. *Journal of Economic Perspectives*, *8*(4), 19–43.

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., Thau, D., Stehman, S. V., Goetz, S. J., Loveland, T. R., Kommareddy, A., Egorov, A., Chini, L., Justice, C. O., & Townshend, J. R. G. (2013). High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science*, *342*(6160), 850–853. https://doi.org/10.1126/science.1244693

Harmsworth, G. R., & Awatere, S. (2013). Indigenous Māori knowledge and perspectives of ecosystems. *Ecosystem Services in New Zealand—Conditions and Trends. Manaaki Whenua Press, Lincoln, New Zealand*, *274*.

Harris, S., Cichota, R., Lilburne, L., & Fraser, C. (2024). Predicting the Productive Potential of Land for Pastoral Sheep and Beef Farming in New Zealand. *Available at SSRN 4827752*. https://papers.ssrn.com/sol3/papers.cfm?abstract_id=4827752

Heinemann, A., & Gray, A. (2024). National Panel Survey of Marine Recreational Fishers 2022–2023. *New Zealand Fisheries Assessment Report*, *51*(116), 50.

Ipsos & Maritime New Zealand. (2024). *Maritime NZ Recreational Boating Monitor: Annual Wrap-up, 2023-24 Financial Year* [Summary report]. Ipsos. https://www.maritimenz.govt.nz/media/0nfbzctv/ipsos-maritimenz-2023-2024-report.pdf

Iversen, A., Asche, F., Hermansen, Ø., & Nystøyl, R. (2020). Production cost and competitiveness in major salmon farming countries 2003–2018. *Aquaculture*, *522*, 735089. https://doi.org/10.1016/j.aquaculture.2020.735089

Jiang, L. (2015). *Economic value of freshwater recreational angling in Otago: A travel cost method approach*.

Johnston, R. J., Boyle, K. J., Adamowicz, W. (Vic), Bennett, J., Brouwer, R., Cameron, T. A., Hanemann, W. M., Hanley, N., Ryan, M., Scarpa, R., Tourangeau, R., & Vossler, C. A. (2017). Contemporary Guidance for Stated Preference Studies. *Journal of the Association of Environmental and Resource Economists*, *4*(2), 319–405. https://doi.org/10.1086/691697

Johnston, R. J., Rolfe, J., Rosenberger, R. S., & Brouwer, R. (2015). Benefit transfer of environmental and resource values. *The Economics of Non-Market Goods and Resources*, *14*.

Kaur, A., Sun, M., & Tsigaris, P. (2025). ASSESSING RECREATIONAL USE VALUE IN BRITISH COLUMBIA'S LARGEST URBAN PARK. *PARKS*.

Kaval, P., & Yao, R. (2007). *New Zealand outdoor recreation benefits* (University of Waikato Working Paper in Economics No. 07/14). University of Waikato.

Kaval, P., & Yao, R. (2010). Research note: A New Zealand outdoor recreation benefit database for benefit transfer studies. *Annals of Leisure Research*, *13*(4), 709–715. https://doi.org/10.1080/11745398.2010.9686872

Kennedy, C. M., Lonsdorf, E., Neel, M. C., Williams, N. M., Ricketts, T. H., Winfree, R., Bommarco, R., Brittain, C., Burley, A. L., Cariveau, D., Carvalheiro, L. G., Chacoff, N. P., Cunningham, S. A., Danforth, B. N., Dudenhöffer, J., Elle, E., Gaines, H. R., Garibaldi, L. A., Gratton, C., … Kremen, C. (2013). A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. *Ecology Letters*, *16*(5), 584–599. https://doi.org/10.1111/ele.12082

Kerr, G. (2003). *Extra-market values and water management in New Zealand*. https://www.academia.edu/download/40311295/Extra-market_values_and_water_management20151123-8497-ty84yr.pdf

Kerr, G. (2019). Himalayan tahr (Hemitragus jemlahicus) recreational hunting values. *Wildlife Research*, *46*(2), 114–126. https://doi.org/https://doi.org/10.1071/WR18027

Kerr, G., & Abell, W. (2014). Big game hunting in New Zealand: Per capita effort, harvest and expenditure in 2011–2012. *New Zealand Journal of Zoology*, *41*(2), 124–138. https://doi.org/10.1080/03014223.2013.870586

Kerr, G., & Latham, N. (2011). New Zealand marine recreational fishing values. *Land Environment and People (LEaP) Research Report*, (29). https://www.academia.edu/download/40311311/New_Zealand_marine_recreational_fishing_20151123-18309-1j06dhz.pdf

Kerr, G. N., & Abell, W. (2016). What are they hunting for? Investigating heterogeneity among sika deer (Cervus nippon) hunters. *Wildlife Research*, *43*(1), 69–79. https://doi.org/https://doi.org/10.1071/WR15117

Kerr, G., & Woods, A. (2010). *New Zealand big game hunting values: A benefit transfer study*. Lincoln University. Faculty of Environment, Society and Design.

Klepeis, N. E., Nelson, W. C., Ott, W. R., Robinson, J. P., Tsang, A. M., Switzer, P., Behar, J. V., Hern, S. C., & Engelmann, W. H. (2001). The National Human Activity Pattern Survey (NHAPS): A resource for assessing exposure to environmental pollutants. *Journal of Exposure Science & Environmental Epidemiology*, *11*(3), 231–252. https://doi.org/10.1038/sj.jea.7500165

Lachowycz, K., & Jones, A. P. (2014). Does walking explain associations between access to greenspace and lower mortality? *Social Science & Medicine*, *107*, 9–17. https://doi.org/10.1016/j.socscimed.2014.02.023

Li, J., & Wang, Y. (2023). Ecosystem services assessment from capacity to flow: A review. *Transactions in Earth, Environment, and Sustainability*, *1*(1), 80–93. https://doi.org/10.1177/2754124X221141991

Maes, J., Liquete, C., Teller, A., Erhard, M., Paracchini, M. L., Barredo, J. I., Grizzetti, B., Cardoso, A., Somma, F., Petersen, J.-E., Meiner, A., Gelabert, E. R., Zal, N., Kristensen, P., Bastrup-Birk, A., Biala, K., Piroddi, C., Egoh, B., Degeorges, P., … Lavalle, C. (2016). An indicator framework for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. *Ecosystem Services*, *17*, 14–23. https://doi.org/10.1016/j.ecoser.2015.10.023

Manley, B. (2021). *Stumpage values across New Zealand* (Nos. 978-1-99-101930–1; Ministry for Primary Industries Information Paper, Issue 2021/06). Ministry for Primary Industries. https://www.mpi.govt.nz/

Marsh, D., & Mkwara, L. (2013). *Review of Freshwater Non-Market Value Studies* [Report prepared for Waikato Regional Council].

Matheson, L., Herzig, A., Muller, C., Stone, C., Dowd, A., Brazendale, R., & Gray, S. (2024). *Steering land use change to meet water quality targets*.

Matthews, Y. S., Holland, P., Matheson, F. E., Craggs, R. J., & Tanner, C. C. (2024). Evaluating the Cost-Effectiveness of Green Infrastructure for Mitigating Diffuse Agricultural Contaminant Losses. *Land*, *13*(6), 748.

Matthews, Y., Scarpa, R., & Marsh, D. (2018). Cumulative attraction and spatial dependence in a destination choice model for beach recreation. *Tourism Management*, *66*, 318–328. https://doi.org/10.1016/j.tourman.2017.12.009

McDougall, M., Nieuwboer, B., & Garrick, A. S. (2018). *Can management regime, vegetative, and morphological characteristics of drains in the Bay of Plenty explain the presence of mallards and their broods?* Fish and Game New Zealand, Eastern Region.

Ministry of Transport. (2023). *Domestic Transport Costs and Charges Study* (Working Paper D4). https://www.transport.govt.nz/assets/Uploads/DTCC-WP-D4-Air-quality-and-GHG-June-2023.pdf

Monaghan, R., Manderson, A., Basher, L., Smith, C., Burger, D., Meenken, E., & McDowell, R. (2021). Quantifying contaminant losses to water from pastoral landuses in New Zealand I. Development of a spatial framework for assessing losses at a farm scale. *New Zealand Journal of Agricultural Research*, *64*(3), 344–364. https://doi.org/10.1080/00288233.2021.1936572

Morris, R., Davis, S., Grelet, G.-A., Doscher, C., & Gregorini, P. (2024). A Model for Spatially Explicit Landscape Configuration and Ecosystem Service Performance, ESMAX: Model Description and Explanation. *Sustainability*, *16*(2), 876. https://doi.org/10.3390/su16020876

Narayan, S., Beck, M. W., Wilson, P., Thomas, C. J., Guerrero, A., Shepard, C. C., Reguero, B. G., Franco, G., Ingram, J. C., & Trespalacios, D. (2017). The value of coastal wetlands for flood damage reduction in the northeastern USA. *Scientific Reports*, *7*(1), 9463.

NCAVES & MAIA. (2022). *Monetary valuation of ecosystem services and ecosystem assets for ecosystem accounting: Interim version, 1st edition* (1st edition). United Nations Department of Economic and Social Affairs, Statistics Division.

New Zealand Ministry for the Environment. (2024). *National Policy Statement for Freshwater Management 2020 – Amended October 2024*. https://environment.govt.nz/assets/publications/Freshwater/NPSFM-amended-october-2024.pdf

New Zealand Treasury. (2025). *Assessing climate change and environmental impacts in the CBAx tool*. The Treasury. https://www.treasury.govt.nz/sites/default/files/2025-11/cbax-tool-climate-environmental-impacts-oct25.pdf

Nugent, G. (1992). Big-game, small-game, and gamebird hunting in New Zealand: Hunting effort, harvest, and expenditure in 1988. *New Zealand Journal of Zoology*, *19*(3–4), 75–90.

OECD. (2020). Nature-Based Solutions for Adapting to Water-Related Climate Risks. *OECD Environment Policy Papers*, 32.

Paracchini, M. L., Zulian, G., Kopperoinen, L., Maes, J., Schägner, J. P., Termansen, M., Zandersen, M., Perez-Soba, M., Scholefield, P. A., & Bidoglio, G. (2014). Mapping cultural ecosystem services: A framework to assess the potential for outdoor recreation across the EU. *Ecological Indicators*, *45*, 371–385.

Pascoe, S. (2019). Recreational beach use values with multiple activities. *Ecological Economics*, *160*, 137–144. https://doi.org/10.1016/j.ecolecon.2019.02.018

Paul, T., Kimberley, M. O., & Beets, P. N. (2021). *Natural forests in New Zealand – a large terrestrial carbon pool in a national state of equilibrium* (p. 34). https://doi.org/10.1186/s40663-021-00312-0

Paul, T., & Wakelin, S. (2023). *Carbon Stocks and Stock Changes in New Zealand's Pre-1990 Natural Forest* (Ministry for the Environment client report 22255). Scion.

Petheram, C., & McMahon, T. A. (2019). Dams, dam costs and damnable cost overruns. *Journal of Hydrology X*, *3*, 100026. https://doi.org/10.1016/j.hydroa.2019.100026

Phillips, C., Basher, L., & Spiekermann, R. (2020). Biophysical performance of erosion and sediment control techniques in New Zealand: A review. *Landcare Research Contract Report LC3761*.

Plummer, M. L. (2009). Assessing benefit transfer for the valuation of ecosystem services. *Frontiers in Ecology and the Environment*, *7*(1), 38–45. https://doi.org/10.1890/080091

Prayaga, P. (2017). Estimating the value of beach recreation for locals in the Great Barrier Reef Marine Park, Australia. *Economic Analysis and Policy*, *53*, 9–18. https://doi.org/10.1016/j.eap.2016.10.001

Ravazdezh, F., & Rivers, N. (2025). Quasi-experimental evidence that the urban tree canopy reduces residential energy consumption. *Energy and Buildings*, *344*, 115992. https://doi.org/10.1016/j.enbuild.2025.115992

Richards, D., Herzig, A., Abbott, M., Ausseil, A.-G., Guo, J., Sood, A., & Lavorel, S. (2023). Diverse contributions of nature to climate change adaptation in an upland landscape. *Ecosystems and People*, *19*(1), 2225647. https://doi.org/10.1080/26395916.2023.2225647

Rojas-Nazar, U. A., Gardner, J. P. A., Rouchon, A., & Cullen, R. (2022). Valuing Marine Reserves: A case study from two locations in central New Zealand. *Marine Policy*, *140*, 105051. https://doi.org/10.1016/j.marpol.2022.105051

Rolfe, J., & Gregg, D. (2012). Valuing beach recreation across a regional area: The Great Barrier Reef in Australia. *Ocean & Coastal Management*, *69*, 282–290.

Rosenberger, R., & Phipps, T. (2007). Correspondence and convergence in benefit transfer accuracy: Meta-analytic review of the literature. *Environmental Value Transfer: Issues and Methods*, 23–43.

Samarasinghe, O. E., & Sharp, B. M. H. (2008). The value of a view: A spatial hedonic analysis. *New Zealand Economic Papers*, *42*(1), 59–78. https://doi.org/10.1080/00779950809544413

Sanders, G. (2024). *Licence Holder Insights*. New Zealand Fish & Game. https://www.fishandgame.org.nz/assets/DMS/About-us/FG-Regional-Councils/North-Canterbury/Council-Downloads/2024/NCant_LHInsights.pdf

Schroeder, H. W. (1982). Preferred features of urban parks and forests. *Journal of Arboricuture 8 (12): 317-322*, *8*(12).

Seppelt, R., Dormann, C. F., Eppink, F. V., Lautenbach, S., & Schmidt, S. (2011). A quantitative review of ecosystem service studies: Approaches, shortcomings and the road ahead. *Journal of Applied Ecology*, *48*(3), 630–636.

Seymour, G. (2020). *Solid Fuel Burner Testing and Standards to be Considered Within Proposed Amendments to National Environmental Standards for Air Quality*. Ministry for the Environment. https://environment.govt.nz/assets/Publications/Files/NESAQ-review-solid-fuel-burner-testing-and-standards.pdf

Sheridan, R. E. (2019). *Environmental and genetic influences on growth, flowering, and nectar production in mānuka (Leptospermum scoparium JR Forst. & G. Forst.): A thesis presented in partial fulfilment of the requirements for the degree of Doctor of Philosophy in Plant Science, School of Agriculture and Environment, Massey University, Palmerston North, New Zealand*.

Soliman, T., & Walsh, P. (2005). Valuing the effect of soil erosion on productivity. *MBIE Programme "Smarter Targeting of Erosion Control (STEC)"*, *7*, 1–14.

Sport New Zealand. (2025). *Active NZ Survey 2024/25*. https://sportnz.org.nz/resources/active-nz-survey-202425/

Sprague, R. I., & Wiser, S. K. (2024). *Investigating a unifying ecosystem typology for all of New Zealand* (Contract Report LC4514). Manaaki Whenua – Landcare Research.

Stange, E. E., Zulian, G., Rusch, G. M., Barton, D. N., & Nowell, M. (2017). Ecosystem services mapping for municipal policy: ESTIMAP and zoning for urban beekeeping. *One Ecosystem*, *2*, e14014.

Stephenson, F., Bulmer, R., Leathwick, J., Brough, T., Clark, D., & Greenfield, B. (2020). Development of a New Zealand seafloor community classification (SCC). *NIWA Report Prepared for Department of Conservation (DOC). Available Online at: Https://Www. Doc. Govt. Nz/Globalassets/Documents/Conservation/Marine-and-Coastal/Marine-Protected-Areas/Development-of-New-Zealandseafloor-Community-Classification. Pdf*.

Stott, R., Sukias, J. P. S., McKergow, L. A., Davies-Colley, R. J., & Tanner, C. C. (2023). Net export of E. coli from a constructed wetland treating agricultural drainage. *Ecological Engineering*, *194*, 107023. https://doi.org/10.1016/j.ecoleng.2023.107023

Subroy, V., Gunawardena, A., Polyakov, M., Pandit, R., & Pannell, D. J. (2019). The worth of wildlife: A meta-analysis of global non-market values of threatened species. *Ecological Economics*, *164*, 106374. https://doi.org/10.1016/j.ecolecon.2019.106374

Swaffield, S. R., & McWilliam, W. J. (2013). Landscape aesthetic experience and ecosystem services. *School of Landscape Architecture, Lincoln University*, (2.6), 349–362.

Tabatabaie, S., Litt, J. S., & Muller, B. H. F. (2023). Sidewalks, trees and shade matter: A visual landscape assessment approach to understanding people's preferences for walking. *Urban Forestry & Urban Greening*, *84*, 127931. https://doi.org/10.1016/j.ufug.2023.127931

Tadaki, M., Sinner, J., & Chan, K. M. A. (2017). Making sense of environmental values: A typology of concepts. *Ecology and Society*, *22*(1), art7. https://doi.org/10.5751/ES-08999-220107

Tait, P., Saunders, C., Miller, S., Rutherford, P., Greer, G., & Abell, W. (2017). *Assessing New Zealand public preferences for native biodiversity outcomes across habitat types: A choice experiment approach incorporating habitat engagement*. AERU, Lincoln University.

Tanner, C. C., Depree, C., Sukias, J. P. S., Wright-Stow, A. E., Burger, D. F., & Goeller, B. (2022). *Constructed Wetland Practitioner Guide: Design and Performance Estimates*. DairyNZ.

Tanner, C., Matthews, Y., Craggs, R., & Holland, P. (2026). Optimizing Cost-Effective Mitigation Strategies for Diffuse Agricultural Pollution Across Diverse New Zealand Landscapes. *Creating Multifunctional Farms and Catchments (Eds C.L.Christensen, D.J.Horne and R.Singh)*, (Occasional Report No. 37).

Thomas, S., Zyskowski, R., & Francis, G. (2011). *Effectiveness of policy options to reduce nitrous oxide emissions and nitrate leaching from arable and annual horticultural land* *uses* (MAF Technical Paper No: 2011/71). https://www.mpi.govt.nz/dmsdocument/2559-Effectiveness-at-a-farmer-level-of-the-use-of-nitrification-inhibitors-and-charges-on-nitrogen-fertilisers

United Fresh New Zealand. (2025). *Fresh Facts 2025*. https://unitedfresh.co.nz/assets/site/Fresh-Facts-2025-%E2%80%93-Online-Version-%E2%80%93-28.10.25.pdf

United Nations. (2022). *Guidelines on Biophysical Modelling for Ecosystem Accounting*. United Nations Department of Economic and Social Affairs, Statistics Division, New York. https://seea.un.org/sites/seea.un.org/files/publications/guidancebiomodelling_v36_30032022_web.pdf

Vymazal, J. (2005). Removal of Enteric Bacteria in Constructed Treatment Wetlands with Emergent Macrophytes: A Review. *Journal of Environmental Science and Health, Part A*, *40*(6–7), 1355–1367. https://doi.org/10.1081/ESE-200055851

Walther, F., Barton, D. N., Schwaab, J., Kato-Huerta, J., Immerzeel, B., Adamescu, M., Andersen, E., Arámbula Coyote, M. V., Arany, I., Balzan, M., Bruggeman, A., Carvalho-Santos, C., Cazacu, C., Geneletti, D., Giuca, R., Inácio, M., Lagabrielle, E., Lange, S., Clec'h, S. L., … Grêt-Regamey, A. (2025). Uncertainties in ecosystem services assessments and their implications for decision support – A semi-systematic literature review. *Ecosystem Services*, *73*, 101714. https://doi.org/10.1016/j.ecoser.2025.101714

Water Safety New Zealand. (2022). *National Coastal and Water Safety Survey in New Zealand 2022*. https://nzsar.govt.nz/assets/Downloadable-Files/National-Coastal-and-Water-Safety-survey-in-NZ-in-2023-July-13-presentation-.pdf

Webb, T., & Willson, A. (1995). *A manual of land characteristics for evaluation of rural land* (Landcare Research Science Series 10). Manaaki Whenua Press. https://soils.landcareresearch.co.nz/assets/Publications/LRSS-10-Manual-land-characteristics.pdf

Weerappulige, A., & Khoo, J. (2020). *Health and active modes impacts: A technical paper prepared for the Investment Decision-Making Framework Review*. Waka Kotahi NZ Transport Agency.

Wilcock, R. J., Müller, K., Van Assema, G. B., Bellingham, M. A., & Ovenden, R. (2012). Attenuation of Nitrogen, Phosphorus and E. coli Inputs from Pasture Runoff to Surface Waters by a Farm Wetland: The Importance of Wetland Shape and Residence Time. *Water, Air, & Soil Pollution*, *223*(2), 499–509. https://doi.org/10.1007/s11270-011-0876-8

Yang, J., Moore, C., Rajanayaka, C., Shiona, H., Hemmings, B., Oluwunmi, P., Cameron, S., & Woodward, S. J. R. (2025). National Nutrient Contribution Dynamics in New Zealand Rivers. *Hydrological Processes*, *39*(5), e70161. https://doi.org/10.1002/hyp.70161

Yao, R., Botero, J., Mkwara, L., Solaymani, S., & Wegner, S. (2023). *Timber and trails: The economic and well-being impacts of recreational mountain biking in New Zealand* (Confidential Final Report Scion contract number QT-9645). Scion.

Zulian, G., Paracchini, M.-L., Maes, J., & Liquete, C. (2013). ESTIMAP: Ecosystem services mapping at European scale. *Publications Office of the European Union, Luxembourg*.