Australian Government
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Measuring and accounting for the benefits of restoring coastal blue carbon ecosystems:

The Guide (Version 1)

Authors

Paul Carnell[1], Kym Whiteoak[2], Vincent Raoult[1], Michael Vardon[3], Maria Fernanda Adame[4], Michael Burton[5], Rod M. Connolly[4], Will Glamore[6], Alice Harrison[6], Jeff Kelleway[7], Catherine E. Lovelock[8], Emily Nicholson[1,13], Melissa Nursey-Bray[9], Celeste Hill[9], Nina Wootton[9], Dewayne Mundraby[10], Dale Mundraby[10], Christopher J. Owers[11], Jacqueline B. Pocklington[1], Abbie Rogers[5], Tafesse Estifanos[5], Fitalew Taye[5], Kerrylee Rogers[7], Matthew D Taylor[12], Emma Asbridge[7], Daniel E. Hewitt[4], Peter I. Macreadie[1]

[1] School of Life and Environmental Sciences, Deakin University, VIC, Australia

[2] Canopy Economics and Policy, Melbourne, VIC, Australia

[3] Fenner School of Environment and Society, The Australian National University, Canberra, ACT, Australia

[4] Coastal and Marine Research Centre, Australian Rivers Institute, School of Environment and Science, Griffith University, Gold Coast, QLD, Australia

[5] Centre for Environmental Economics and Policy, UWA School of Agriculture and Environment, The University of Western Australia, Crawley, WA, Australia

[6] Water Research Laboratory, School of Civil and Environmental Engineering, University of New South Wales, Sydney, NSW, Australia

[7] School of Earth, Atmospheric and Life Science, GeoQuEST Research Centre, University of Wollongong, Wollongong, NSW, Australia

[8] School of Biological Sciences, The University of Queensland, St Lucia, QLD, Australia

[9] Geography, Environment, Population, University of Adelaide, Adelaide, SA, Australia

[10] Mandingalbay Yidinji Aboriginal Corporation, QLD, Australia

[11] School of Environmental and Life Sciences, College of Engineering, Science and Environment, University of Newcastle, Callaghan, NSW, Australia

[12] Port Stephens Fisheries Institute, New South Wales Department of Primary Industries, Taylors Beach, NSW, Australia

[13] School of Agriculture, Food and Ecosystem Sciences, Faculty of Science, The University of Melbourne, VIC, Australia

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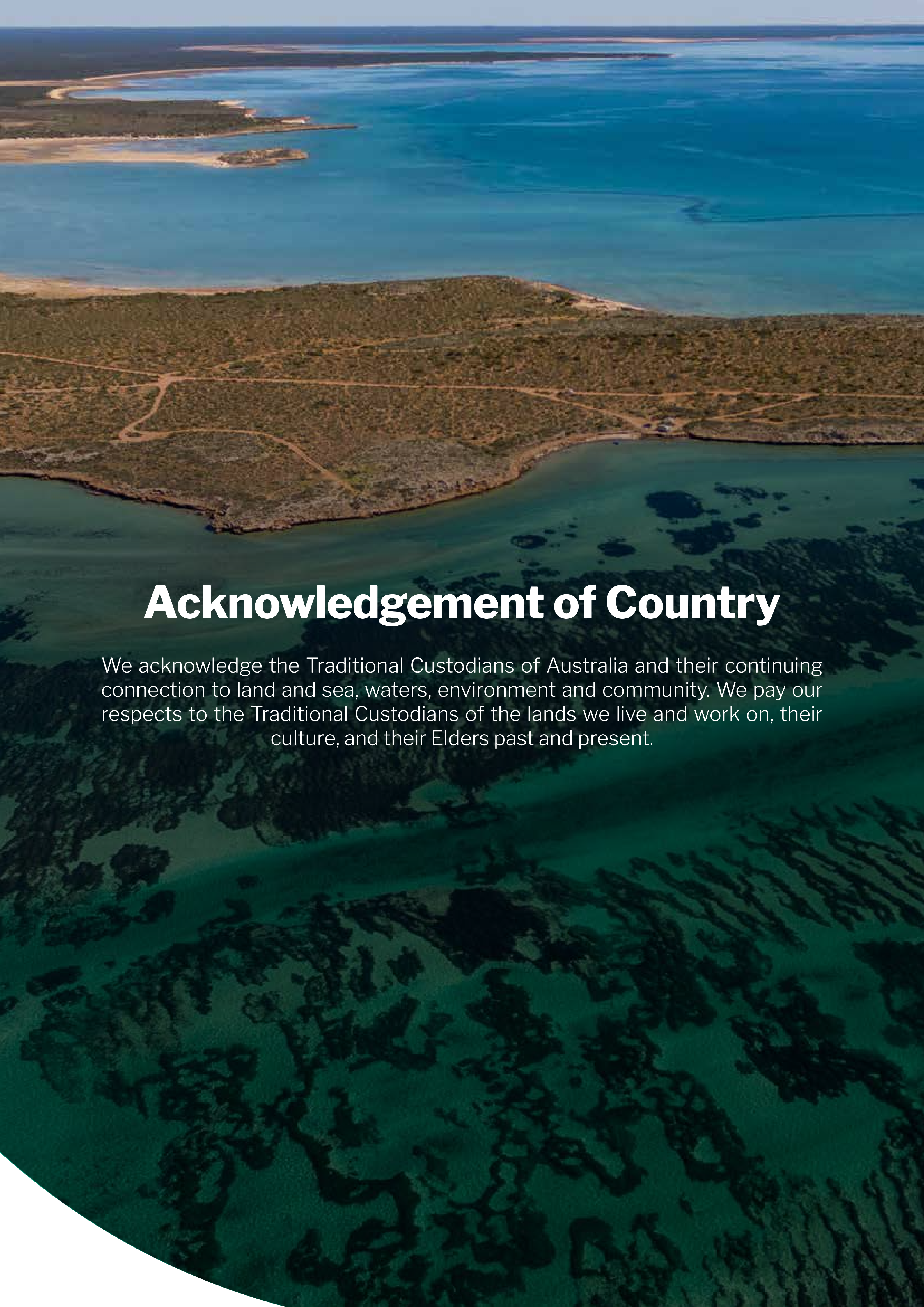

Acknowledgement of Country

We acknowledge the Traditional Custodians of Australia and their continuing connection to land and sea, waters, environment and community. We pay our respects to the Traditional Custodians of the lands we live and work on, their culture, and their Elders past and present.

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Executive Summary

This Guide to Measuring and Accounting for the Benefits of Restoring Coastal Blue Carbon Ecosystems – Version 1 (hereafter 'the Guide') introduces a process for assessing restoration benefits of coastal blue carbon ecosystems, as well as providing detailed advice on measuring these benefits, and presenting the results in a format aligned with formal environmental economic accounting principles. The project was commissioned by the Department of Climate Change, Energy, the Environment and Water (DCCEEW) and delivered by a consortium led by Deakin University.

This Guide is a 'working version' which will be tested and refined. This includes further exploration of the application of the UN SEEA (United Nations System of Environmental Economic Accounting) framework at a project level, to understand how it may be used to measure and account for the impacts of restoration activities on the economic, environmental and social value of coastal blue carbon ecosystems.

Coastal ecosystems (such as mangroves, tidal marshes, and seagrasses) are highly productive and are home to unique and diverse plant and animal species. They are recognised for their contribution to climate change mitigation and adaptation, protection from storm surge and sea level rise, erosion prevention along shorelines, coastal water quality regulation, nutrient cycling, sediment trapping, habitat and food provision for commercial, recreational and endangered marine species, as well as recreational and traditional owner values. Due to their ability to sequester and store large amounts of carbon, they are referred to as 'blue carbon ecosystems'.

Australia's coastal blue carbon ecosystems are in continued decline due to a range of threats, and interest is growing from private and public sources to actively restore them. The Australian Government commissioned this study to develop a process by experts for measuring and accounting for the likely benefits and ecosystem services resulting from restoring blue carbon ecosystems.

What this Guide is for

This Guide is designed to be used by a range of stakeholders including practitioners planning to implement a coastal blue carbon ecosystem restoration project and report on the impacts in physical and monetary terms, and project funders and managers seeking to understand the potential outcomes of a restoration project and how they might be measured and reported on. The Guide provides the following information and guidance:

- Methods for measuring various parameters that can be used to assess the benefits for climate, ecosystems, and people from a coastal blue carbon ecosystem project.
- Information about Environmental Economic Accounting (EEA) and the United Nations System of Environmental Economic Accounting (UN SEEA). A process that can be used to scope an assessment, design reporting accounts, identify and assemble data, compile, and then report results.
- Detailed guidance on how to assemble information about a broad range of project outcomes in physical and monetary terms, including recommended methodologies, available data, and key considerations.
- An example set of tables that can be used to report on project impacts in a manner that aligns with the UN SEEA Ecosystem Accounting (SEEA-EA).

It is expected that users of this Guide will tailor the advice provided to the specific needs of their restoration project, reflecting context, priority outcomes, available resources for field and other data collection, and the purpose of their individual assessment. Where possible, the Guide provides measurement methodologies reflecting higher and lower budgets and data availability to allow the guide to be used for projects at different scales.

As such, the methodologies described in the Guide are not intended to be followed like an instruction manual – they are proposed by relevant experts as defensible methodologies to report on likely restoration benefits, however their use will be guided by proponent needs and the context of their own assessments. This Guide is also intended to assist those measuring possible benefits of a restoration project – it does not provide guidance on how to prepare for and implement a restoration project itself.

How to use this Guide

The proposed process for implementing this Guide is presented in Figure 1.1. Users are recommended to scope and frame their project, considering anticipated restoration project benefits of most interest to inform data collection needs, and aligning these with proposed reporting arrangements.

At this foundation stage, users of the Guide should consider the range of skillsets that will be needed to deliver on the assessment, and the key stakeholders that will be valuable to engage with. Following this, users should establish baseline information on ecosystem extent and condition that will be used to measure expected benefits over time. Detailed methodologies are provided on how to do this for coastal blue carbon ecosystems.

Drawing upon this baseline ecosystem data, this Guide offers detailed methodologies for assessing restoration effects including for climate, ecosystems and people. Users are expected to choose a combination of these that best suits their project context, assessment objectives and project budget. The Guide provides users with low and high-cost methodologies where appropriate and available, to measure restoration benefits and to present this data, if desired, in physical and monetary terms using the SEEA framework.

Detailed guidance is provided on measuring a broad range of outcomes from the restoration project: ecosystem condition including numerous biodiversity-related outcomes, carbon sequestration and emission reduction, water purification, coastal protection, fish production, Traditional Owner cultural values, as well as other cultural services such as recreation, and community (existence) values for restoration outcomes.

Lastly, the Guide provides an example set of SEEA-EA aligned accounts. Users can draw on these tables should they want to prepare accounting tables representing their restoration project area and the changes over time from restoration actions.

In addition to this document, the project team has applied the Guide to two site-level case studies: the Hunter River Estuary in New South Wales and East Trinity Inlet in Cairns. Detailed project reports are provided separately on the process used to assess the benefits of these restoration projects, methodologies employed, and outputs produced including presentation of SEEA-EA aligned ecosystem accounts.

1. Introduction

1.1 Guide overview and purpose

Coastal ecosystems are critical to preserve human well-being and global biodiversity. Mangroves, tidal marshes, and seagrasses are among the most productive ecosystems globally, and this drives a unique and diverse ecosystem which supports a variety of species[1]. Both the physical presence of the plants themselves and the diverse array of species provide services and benefits essential for the well-being of coastal communities[2] .

These coastal ecosystems are also important for climate adaptation and resilience along coasts, including protection from storm surge and sea level rise, preventing erosion along shorelines, regulating coastal water quality, nutrient recycling, sediment trapping, habitat provision for many commercially important and endangered marine species, and food security for many coastal communities around the world[3].

In addition, coastal ecosystems such as mangroves, tidal marshes, seagrass meadows and supratidal swamp forests help mitigate climate change by sequestering and storing large amounts of carbon, known as blue carbon, from the atmosphere and oceans[4]. For simplicity, these ecosystems are referred to as blue carbon ecosystems and this term is used throughout.

Blue carbon ecosystems have declined in extent and condition over recent decades, in Australia and internationally[5]. Greater awareness of this loss, along with growing understanding of the values provided by these ecosystems, has led to greater interest in the restoration of blue carbon ecosystems. Restoration here refers to changing the degraded ecosystem so that it once again provides some or all the ecosystem services and functions lost as a result of human

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[1] Sievers, M., et al. (2019). The Role of Vegetated Coastal Wetlands for Marine Megafauna Conservation. Trends in Ecology & Evolution, 34(9), 807-817. https://doi.org/10.1016/j.tree.2019.04.004

[2] Barbier, E. B., et al. (2011). The value of estuarine and coastal ecosystem services. Ecological Monographs, 81(2), 169–193. https://doi.org/10.1890/10-1510.1

[3] Himes-Cornell, A., et al. (2018). Valuing ecosystem services from blue forests: A systematic review of the valuation of salt marshes, sea grass beds and mangrove forests. Ecosystem Services, 30(Part A), 36-48. https://doi.org/10.1016/j.ecoser.2018.01.006

[4] McLeod, E., et al. (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Frontiers in Ecology and the Environment, 9(10), 552-560. https://doi.org/10.1890/110004

[5] For examples see Department of the Environment and Energy. (2017). National Inventory Report 2015 Volume 2, Commonwealth of Australia 2017; OR Macreadie, P. I., et al. (2017). Carbon sequestration by Australian tidal marshes. Scientific Reports, 7, 44071. https://doi.org/10.1038/srep44071; OR Statton, J., et al. (2018). Decline and Restoration Ecology of Australian Seagrasses. In A. Larkum, G. Kendrick, & P. Ralph (Eds.), Seagrasses of Australia (pp. 481-504). Springer, Cham. https://doi.org/10.1007/978-3-319-71354-0_20

activities. Restoration of blue carbon ecosystems typically involves return of tidal inundation and/or excluding livestock through fence installation. The approaches for restoring these ecosystems have been tested in many locations and is relatively well understood: the principal barrier to large-scale restoration of blue carbon ecosystems is a lack of investment.

Quantifying and valuing the benefits of restoration for climate, people and ecosystems is one way to drive further investment into blue carbon ecosystem restoration. However, as yet there are no common standards for the best ways to measure and verify the diverse benefits of restoring coastal blue carbon ecosystems. Diverse methods are being applied across Australia and the world, often at significant cost and requiring considerable specialist expertise that would be unaffordable or impractical for lower budget projects.

The System of Environmental Economic Accounting - Ecosystem Accounting (SEEA-EA) is an international standard to structure information and track the changes in ecosystem extent, condition, and the ecosystem services that benefit society. SEEA-EA is an organising framework for collecting and reporting data on physical and economic aspects of the environment. It is increasingly being used to record data on the natural value of marine and coastal environments, the services these ecosystems provide humans, and the impact/pressures of economic and other human activity on the environment. For example, in August 2022 the Australian Bureau of Statistics released the first phase of the National Ocean Ecosystem Accounts[6] and in November 2020, an ocean accounting pilot project was completed for Geographe Bay in Western Australia[7].

At project level, the application of SEEA-EA to coastal blue carbon ecosystems is in its infancy, with few experimental case studies. Some case studies have considered services and benefits of coastal wetlands to people and nature; this includes research on services such as wave attenuation[8], their carbon sequestration capacity[9], and fisheries[10]. However, there have been fewer studies that have included coastal wetlands into an SEEA-EA framework[11].

This guide is designed for those seeking to measure the outcomes of restoration projects in coastal blue carbon ecosystems, and provides detailed methodologies on how to calculate these different outcomes in physical and monetary terms, and a proposed framework for presenting outputs in a format aligned to the SEEA–EA, with a view for ongoing improvement of the framework.

1.2 Guide scope and objectives

This Guide provides information on how to identify, measure and report on the benefits of restoring coastal wetlands to the environment, climate and people. This Guide is a 'working version' which will be tested and refined. This includes further exploration of the application of the UN SEEA framework at a project level, to understand how it may be used to measure and account for the impacts of restoration activities on the economic, environmental and social value of coastal blue carbon ecosystems.

This Guide describes an overall process of scoping and implementing an assessment of the outcomes of a coastal blue carbon restoration project, assembling relevant data, developing

[6] Australian Bureau of Statistics. (2022). National Ocean Account, Experimental Estimates. ABS. https://www.abs.gov.au/statistics/environment/environmental-management/national-ocean-account-experimental-estimates/aug-2022

[7] For examples see Department of Climate Change, Energy, the Environment and Water (2020). Ocean accounting pilot for Geographe Marine Park. DCCEEW. https://eea.environment.gov.au/accounts/ocean-accounts/geographe-marine-park; OR World Bank. (2021). The Changing Wealth of Nations 2021: Managing Assets for the Future (see Chapter 6). Washington, DC: World Bank. http://hdl.handle.net/10986/36400; OR World Bank. (2016). Managing Coasts with Natural Solutions: Guidelines for Measuring and Valuing the Coastal Protection Services of Mangroves and Coral Reefs. M. W. Beck and G-M. Lange (Eds.). Wealth Accounting and the Valuation of Ecosystem Services Partnership (WAVES), World Bank, Washington, DC.

[8] Losada et al. 2017; http://hdl.handle.net/10986/27666

[9] Macreadie et al. 2021; https://www.nature.com/articles/s43017-021-00224-1

[10] Jänes et al. 2019; https://onlinelibrary.wiley.com/doi/abs/10.1111/faf.12416

[11] For an example, please see Carnell, P. E., et al. (2019). Mapping Ocean Wealth Australia: The value of coastal wetlands to people and nature. The Nature Conservancy, Melbourne.

a set of environmental economic accounts, as well as providing specific advice for measuring aspects of blue carbon ecosystems that are used to attribute monetary value.

Readers of this Guide should be informed of how to undertake the overall process and develop a working understanding of its implementation for measuring and showcasing the benefits of restoring coastal blue carbon ecosystems. This Guide contains sufficient information to understand the process, options, and key issues, with references to more technical information that can be used by practitioners in restoration project benefits measurement and accounting.

The methodological sections of this report provide brief overviews followed by detailed technical advice that can be used by practitioners to guide the assessment of specific restoration outcomes, such as measuring changes in ecosystem extent and condition, and measuring the value of increased recreational fishing.

The key objectives of the Guide are:

- To provide readers with detailed technical advice on recommended methodologies for assessing key impacts of a restoration project in these ecosystems.
- To inform readers of a process for scoping and implementing an assessment of restoration outcomes over time, and reporting on those restoration outcomes using the SEEA-EA.
- To provide readers with links to additional resources that may be useful in implementing an assessment.
- To assist readers in selecting the correct economic tools for their decision-making needs.

As such, this guide is not a technical handbook but rather a guidance document giving practical advice on how to measure and account for the benefits of restoring coastal blue carbon ecosystems.

Future work could include further exploration of the application of the SEEA-EA framework to project-level coastal blue carbon ecosystem restoration projects, to measure and report on the ecosystem services attributable to a range of restoration activities. There is also scope to develop, refine and propose methods on data collection and account design aligning with the SEEA-EA to create environmental economic accounts to report and track the changes in blue carbon ecosystems resulting from restoration efforts.

1.3 How to use this Guide

The proposed process for implementing this Guide is presented in Figure 1.1. It starts with some foundational steps. Then it provides some guidance on steps that will be required regardless of the framework used, like scoping and framing, data collection, consideration of skillsets required to implement an assessment, and stakeholder engagement.

Following this, the detailed methodological sections begin with detailed advice on how to establish baseline information on ecosystem extent and condition within the project site. This data will be drawn on in assessments of the ecosystem services pre-intervention and will be revisited over time when assessing changes to these in future.

Detailed guidance is provided on measuring a broad range of outcomes from a coastal blue carbon restoration project including those relating to climate, ecosystems and people. As reflected in Figure 1.1 these include: carbon sequestration and emission reduction, water purification, coastal protection, fish production, Traditional Owner cultural values, as well as other cultural services such as recreation, and community (existence) values for restoration and conservation outcomes. The guide also recommends collecting information on the costs of implementing the restoration project, which can be used to assess the cost-effectiveness of the project, and the relative merits of different components.

Lastly, the Guide provides example tables designed to align with the SEEA-EA. Users can draw on these tables should they want to prepare accounting tables representing their restoration project area and the impacts over time of restoration actions.

This report contains summaries of methodologies used for measuring different aspects of restoration projects in coastal blue carbon ecosystems. It is not expected that every proponent will implement every aspect of this Guide. Rather, proponents may apply the framework proposed and draw on the components that are relevant and are prioritised for their particular project.

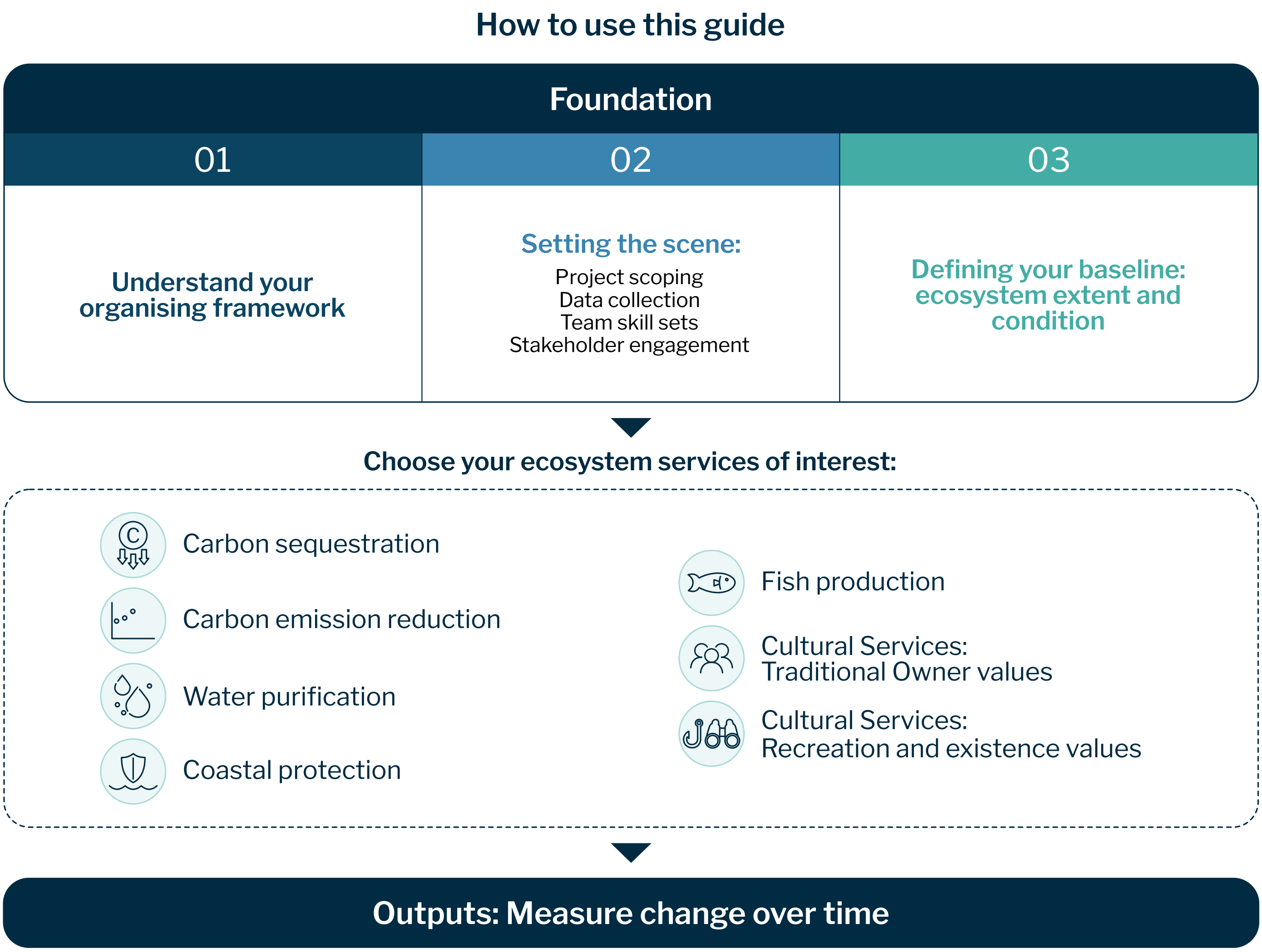


Figure 1.1: How to use the Guide.

The methodologies included in this Guide do not form an official standard – they have been developed by expert practitioners of each component and are recommended as rigorous and defensible methodologies to use in the coastal blue carbon environment with available Australian data sets.

Experts and practitioners who wish to put the guide into practice should read the detailed methodologies presented in A Guide to Measuring and Accounting for the Benefits of Restoring Coastal Blue Carbon Ecosystems: Detailed Methodologies.

In addition to these documents, the project team has applied the Guide to two site-level case studies: the Hunter River Estuary in New South Wales and East Trinity Inlet in Cairns. Detailed project reports are provided separately on the process used to assess the impacts of these restoration projects, methodologies employed, and outputs produced including presentation of tables in SEEA-EA aligned format.

1.4 Structure of this Guide

The remainder of this report is structured in the following way:

- The remainder of this introductory part of the report discusses different ways to measure restoration project outcomes, and proposes a process for undertaking an assessment.
- It presents summaries of the main methodological content of the report. These include:
    - Extent
    - Condition
    - Ecosystem services

1.5 Structure of this Guide: detailed methodologies

This detailed second part of the Guide is directed at practitioners and experts interested in the detailed methodologies for each account. It is expected that users of the detailed section have read the first part of the Guide or that they have prior understanding of SEEA-EA and valuation of ecosystem services.

- In Foundational Data, the Guide explores ecosystem extent and condition in detail.
- In Ecosystem Services, the Guide then goes into detail on the range of ecosystem services users could measure in a coastal blue carbon ecosystem restoration project.
- In the Environmental Protection Accounts section, the Guide explores restoration activities in physical and also monetary accounts.
- In the final part of the report, the Guide proposes a range of tables for presenting data in a format that aligns with the SEEA-EA.

The Guide also provides an appendix with some additional resources, and a glossary of terms.

2. Environmental Economic Accounting

2.1 Overview of Environmental Economic Accounting

At a conceptual level, Environmental Economic Accounting (EEA) is a framework used to compile statistics linking environmental statistics to economic statistics. It provides a rigorous and comprehensive framework of compiling environmental and economic information that can be applied at various scales, including at a project scale. It can be used to demonstrate the broader benefits of biophysical improvements that a restoration project produces over time. EEA is increasingly being used to understand our impact on the marine environment through ocean accounts[12].

Accounting processes record information on stocks (items of value) and flows (supply and use of stocks) related to a particular entity in a systematic way. In business accounting the entity is a business, while in national accounting the entity is a country, but the process is broadly the same. All the transactions into and out of the entity are estimated and recorded and at the end of the period a balance sheet is provided, showing changes in the value of assets over that time.

Ecosystem accounting adds ecosystems into an accounting process by expanding the concept of assets to include ecosystems within it, taking the role of a business in an accounting framework. In this, ecosystems are transacting entities supplying ecosystem services to the economic entities (the flows). Ecosystem accounting also records measures of the extent and condition of ecosystems and the causes for changes in ecosystem extent and condition over the accounting period to forecast changes in accounts into the future, and how making changes to the ecosystem can change these trajectories.

EEA is a useful way of recording and demonstrating the broader benefits of blue-carbon ecosystem restoration projects in a rigorous, repeatable, and comprehensive way. Results can be presented consistently with internationally recognised conventions.

A formal framework for EEA containing internationally agreed standard concepts, definitions, classifications, rules, and tables is the System of Environmental Economic Accounting (SEEA) framework.

[12] Munk Hansen, R., et al. (2021). Ocean accounts inform evidence-based sustainable development of the ocean economy. United Nations Economic and Social Commission for Asia and the Pacific. https://www.unescap.org/blog/ocean-accounts-inform-evidence-based-sustainable-development-ocean-economy

The SEEA framework

The System of Environmental-Economic Accounting (SEEA) has many parts. The SEEA includes the SEEA Central Framework[13] and the SEEA-EA (Ecosystem Accounting)[14], both of which complement the System of National Accounts (SNA)[15]. The SEEA-EA is a spatially based, integrated statistical framework for organising information about ecosystems, tracking changes in ecosystem extent and condition, measuring and valuing ecosystem services and assets and linking this information to measures of economic and human activity. Its focus is to make visible the contributions of nature to the economy and people and the impacts of people and the economy on nature. The main benefit of SEEA-EA relative to the central framework is the spatial diversity that ecosystem accounting can capture. In practice, that means that SEEA-EA allows the use of maps as well as accounting tables, capturing spatial dimensions of stocks.

SEEA-EA applies the System of National Accounts accounting principles to produce a suite of five interlinked accounts, covering the stocks (ecosystem assets) and flows (ecosystem services) related to a particular ecosystem (Figure 2.1). Accounts within the system are presented using biophysical (e.g. hectares, litres – the light circles in Figure 2.1) or monetary measures (e.g. AUD$ - the dark circles in Figure 2.1). Integration between accounts is achieved via the use of common concepts, classifications, and units.

There are two main types of accounts in the System of National Accounts: 1) asset accounts (extent and condition in SEEA-EA) and 2) supply (ecosystem services in SEEA-EA) and use tables (see Section 16 for example tables in a coastal blue carbon environment context). In the SEEA-EA, supply and use tables are accounting tables structured to record flows of goods and services, including ecosystem services, between economic units and the environment, including ecosystems. In supply and use tables, supply always equals use. Asset accounts record the opening and closing stocks between two time periods (e.g. beginning and end of a (Australian) financial year the 1 July and 30 June), and in a SEEA-EA framework asset accounts often focus on ecosystem extent and condition. The difference between the opening and closing stocks must all be accounted for using the reason for change (e.g. due to human or natural factors, termed 'managed' and 'unmanaged' change in SEEA-EA).

See Appendix 1: Additional resources for more detail on concepts and definitions of the SNA and SEEA-EA.

2.2 Introduction to measurement and valuation of ecosystem services

In the SEEA-EA framework, ecosystem services are the connection between ecosystem assets and the production and consumption activity of businesses, households, and governments. SEEA-EA also accounts for services that contribute to non-Systems of National Accounts benefits such as air filtration. Thus, the measurement of ecosystem services in their physical and monetary terms is key to a set of ecosystem accounts[17].

Ecosystem services are defined in the SEEA-EA as: "the contributions of ecosystems to the benefits that are used in economic and other human activity"[12]. This definition includes direct use of services (e.g. Fishing in a mangrove forest), passive enjoyment (eg. The aesthetic enjoyment of viewing a seascape) and indirect receipt of services (e.g. That saltmarsh produced a fish later caught elsewhere). This definition also includes

[13] United Nations, et al. (2017). System of Environmental-Economic Accounting 2012: Central Framework. Manuals & Guides. United Nations. https://doi.org/10.5089/9789211615630.069

[14] United Nations, et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

[15] United Nations, et al. (2009). System of National Accounts, 2008. United Nations Statistical Commission. https://unstats.un.org/unsd/nationalaccount/sna2008.asp

[16] United Nations, et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

[17] Source: after SEEA-EA, UN et al. 2021, p. 32; https://seea.un.org/sites/seea.un.org/files/documents/EA/seea_ea_white_cover_final.pdf

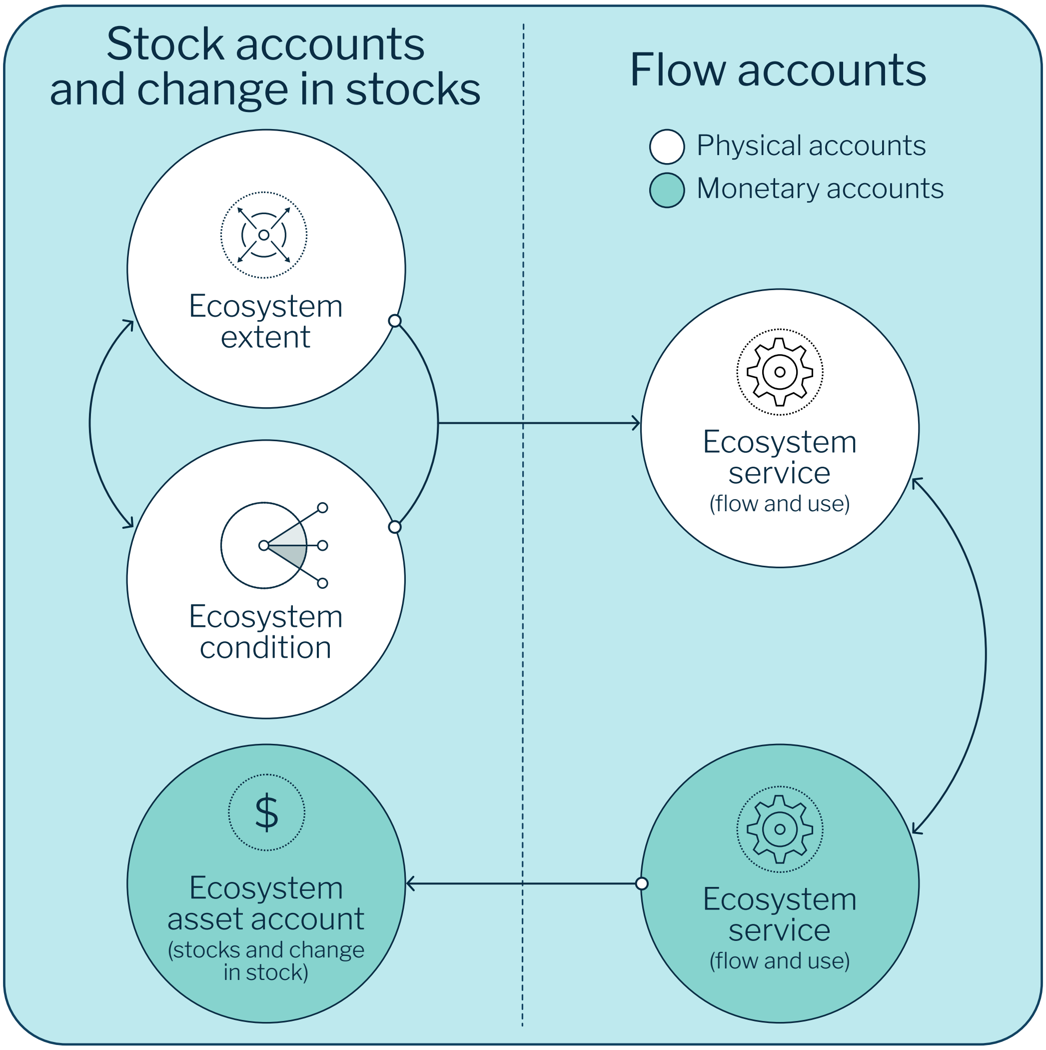


Figure 2.1: Types of ecosystem accounts and their connections[17]. Arrows represent one account being used to determine the other. Linked accounts (closed circles) both impact flow accounts independently.

all forms of interactions both those in person and remote (e.g. footage of a seagrass meadow in a documentary). This means that ecosystem services extend beyond marketed goods, such as fish, and include regulating and maintenance services (e.g. water purification, global climate regulation) and cultural services (e.g. recreation-related services).

Commonly, these types of services are supplied to communities outside markets and because of this their value has often been underappreciated. The focus of accounting for ecosystem services is to describe the range of services, the supply of services by ecosystems, and the users or beneficiaries of these services. This information can be compared between degraded ecosystems and restored ecosystems to understand the impacts of restoration activities on ecosystem services[18].

The key concepts of the ecosystem accounting framework related to ecosystem services concern (i) the supply of ecosystem services to users; and (ii) the contribution of ecosystem services to benefits (i.e., the goods and services ultimately used and enjoyed by people and society). In ecosystem accounting, ecosystem services are recorded as flows between ecosystem assets and economic units; where economic units encompass the various institutional types included in the national accounts, such as businesses, governments, and households. Flows of ecosystem services can be in direct physical flows such as fisheries, but also includes the indirect (or non-use) ecosystem services, such as soil quality regulation services.

[18] United Nations, et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

The relationship between the supply of ecosystem services and the use of ecosystem services will not always be from one ecosystem asset to one economic unit or user. Firstly, ecosystem services can be classified as 'intermediate' or 'final'. In SEEA-EA framework, final ecosystem services are defined as those that are used by economic units, such as global climate regulation services. Intermediate ecosystem services, which are also called 'supporting services', include intra- and inter-ecosystem flows that are not used by economic units. For example, where the intermediate nursery population service is supplied by seagrass meadows, are then an input to the supply of fish biomass provisioning services (final service), which in turn contribute to the benefit of marketed fish. In this case, the nursery population service is treated as intermediate while the biomass provisioning service is final. In other cases, ecosystem services will be supplied through a combination of ecosystem assets (intermediate service). For example, flood control services involving a range of ecosystem types within a catchment. Finally, one ecosystem service can be used by different economic units. For example, air filtration services will contribute to benefits used by both households and businesses.

To measure and value ecosystem services, it is important to understand each service. The below definitions are from the SEEA–EA guidelines.

- ***Provisioning services*** *are those ecosystem services representing the contributions to benefits that are extracted or harvested from ecosystems.*
- ***Regulating and maintenance services*** *are those ecosystem services resulting from the ability of ecosystems to regulate biological processes and to influence climate, hydrological and biochemical cycles, and thereby maintain environmental conditions beneficial to individuals and society.*
- ***Cultural services**** *are the experiential and intangible services related to the perceived or actual qualities of ecosystems whose existence and functioning contributes to a range of cultural benefits.*

* The label "cultural services" is a pragmatic choice and reflects its longstanding use in the ecosystem services measurement community. It is not implied that culture itself is a service, rather it is a summary label intended to capture the variety of ways in which people connect to, and identify with, nature and the variety of motivations for these connections.

The measurement of cultural services includes Indigenous and non-Indigenous values, but while it is important to distinguish between Indigenous and non-Indigenous culture, this report acknowledges that the cultural services and values people attribute to ecosystems is variable, spatially differentiated, and relational[19]. In this context, the evaluation of cultural services needs to map how people value and ascribe meaning to places and recognise that these interpretations may differ widely depending on cultural orientation and affiliation. As outlined by the UK National Ecosystem Assessment, relationality includes three interlinking elements (i) practices (actions people take or things people do), (ii) spaces (settings in which actions happen - places, landscapes or ecosystems) and (iii) ecosystem benefits accruing from the intersection of space and people (meanings or significance generated through specific practices in specific places[20].

Use and non-use values

From an economic perspective, the values people derive form the environment can be characterised as being either "use" or "non-use" values, as described in the Total Economic Value framework[21]. Use values, i.e. are defined as "the benefit to people is revealed through their direct, personal interaction with the environment

[19] Clarke, B., et al. (2021) Integration Cultural Ecosystem Services valuation into coastal wetlands restoration: A case study from South Australia, Environmental Science and Policy, 126, 220 - 229. https://doi.org/10.1016/j.envsci.2020.11.014

[20] Clarke, B., et al. (2021) Integration Cultural Ecosystem Services valuation into coastal wetlands restoration: A case study from South Australia, Environmental Science and Policy, 126, 220 - 229. https://doi.org/10.1016/j.envsci.2020.11.014

[21] Pearce, D. W., and Turner, R. K. (1990). Economics of Natural Resources and the Environment. New York: Harvester Wheatsheaf, 378

(e.g. fishing, swimming at the beach, benefitting from cleaner water), or through indirect use" (e.g.saltmarsh producing a fish caught elsewhere). Incorporation of use values is the focus of SEEA-EA and is relatively straightforward to include into an accounting framework.

Conversely, incorporating non-use values into accounts requires additional considerations. Non-use values are defined as "*those values that people assign to ecosystems (including the associated biodiversity), irrespective of whether they use (directly or indirectly), or intend to use, the ecosystems*". There are two types of non-use values: **1) existence value** is the value is based on the knowledge that an ecosystem is present and **2) bequest value** where the value is based on ensuring the ecosystem is available to future generations. (SEEA-EA para 6.70).

2.3 Measuring physical ecosystem services

In terms of measuring ecosystems services in the SEEA-EA, this is split into the categories of physical and monetary ecosystem services. Measurements of physical ecosystem services (such as tonnes of fish, km's of coastline protected or visits to a coastal wetland) are recorded in accounting tables to show the flow of ecosystem services over an accounting period between ecosystems and users. Measuring physical ecosystem services is often focuses on ecosystem properties and functions; (e.g. rates of denitrification to improve water quality), but also includes the use of ecosystem services (eg. Birdwatching in coastal wetland). A key step in accounting for ecosystem services is linking the supply of services from ecosystems to the people, industries or government that use them. In the SEEA-EA this is achieved through "supply and use" tables, which denote which ecosystems supply the services and then who are the users of these services (commonly split into Households, Government and Industry). Two key components of supply and use tables are: 1) the supply of ecosystem services must equal the use and 2) supply and use need to be recorded in the same unit (eg. tonnes of fish supplied by ecosystems and tonnes of fish caught by commercial fishers).

Integrating mapping data with accounting tables

For some ecosystem services there may be considerable ecosystem type and spatial variation in supply and use of ecosystem services, which may then be possible to display using maps. This approach can make information easier to digest for those less familiar with accounting principles. If mapping is possible for multiple services, overlaying maps for the different ecosystem services may then highlight areas considered ecosystem service "hot spots". Where mapping approaches are used, this same information will still be presented and summarised appropriately in accounting tables, which will be an aggregation of this data from finer scales. Therefore, maps and accounting tables are complementary outputs, simply presenting the same underlying data in different formats.

Exports and Imports of services outside of Ecosystem Accounting Area

The measurement scope of physical and monetary ecosystem service accounts will be established based on the Ecosystem Accounting Area (EA-Area) defined, in this instance the restoration area. However, given that in most restoration sites, users of the services are based outside of the EA Area, means that use by non-resident economic units will need to be recorded in accounting tables. For example, recreational birdwatchers will be travelling to the restoration site from a nearby town. Conversely, imports of ecosystem services supplied by ecosystem assets outside the EA-Area may also be recorded. Entries are made in the final column of the supply table. See section 7.2.6 in the SEEA-EA guidelines for additional discussion on the recording of imports and exports of ecosystem services.

Recording cultural services

The measurement of cultural services generally focused on the type, number and/or quality of an interaction between people and ecosystems. For example, recreation-related services such as fishing and birdwatching are commonly quantified using the number of visits or time spent visiting a specific location/ecosystem. These measures are not a quantification of the supply by an ecosystem, rather the use, they are considered a suitable proxy. Often these measures can be improved by considering specific features and characteristics of the ecosystem (e.g. a site with a higher diversity of bird species will be more attractive to bird watchers). In addition, there are often businesses involved in facilitating and supporting cultural services. Businesses can be involved to either **1) supply access to the ecosystem and/or facilitate activities/experiences within the ecosystem** (e.g. covering entry fees, guides, tour operators, etc.); or **2) supply goods and services to visitors to support their travel to, and time at, an ecosystem** (e.g. hotels, restaurants, transport companies, fuel suppliers) (SEEA-EA para 7.50).

2.4 Monetary valuation

Benefits are the goods and services that are used and enjoyed by people and society. As applied in ecosystem accounting, a benefit will reflect a gain or positive contribution to well-being arising from the use of ecosystem services. As it is possible to measure ecosystem services in physical terms, so too it is possible to measure them in monetary terms, which allows us to understand the total scale of value produced by a restoration activity, as well as understand the relative scale of the many ecosystem services that a project produces. Ecosystem accounting enables measuring the services that underpin the benefits provided to society using physical and monetary benefits.

There are several reasons why people desire to estimate the monetary value of the environment's contribution to the economy and people. In ecosystem accounting, "*the primary motivation for monetary valuation using a common monetary unit is to be able to make comparisons of different ecosystems and/or services that are consistent with measures of products and assets in the national accounts* (SEEA-EA para 8.2)". To achieve this alignment with the national accounts requires the use of **exchange values**. ***Exchange values*** *are the values at which goods, services, labour or assets are in fact exchanged or else could be exchanged for cash* (2008 SNA, para. 3.118). The use of exchange values *facilitates the description of an integrated system of prices and quantities for the economy and the environment that is a core motivation of the SEEA-EA* (SEEA-EA para 8.2). There are a number of methods to measure exchange values, however for most entries in the national accounts, this is done from observed transactions involving market prices. *Market prices are defined as amounts of money that willing buyers pay to acquire something from willing sellers* (2008 SNA, para. 3.119). However, monetary valuation will not be appropriate in all decision-making contexts and it may be more appropriate to use the measurements of physical services instead.

However, by focusing valuation solely on exchange values, the SEEA-EA "recognises that this provides monetary values that exclude welfare measures that may be commonly included in monetary values of the environment used in other contexts". While the exchange valuation approach aligns with national accounts and thus with macro-economic policy, this excludes other valuation approaches such as **welfare values** that are more aligned with micro-economic policy, such as for cost-benefit analysis. **Welfare values** are "those monetary values reflecting the total benefit accruing to consumers and suppliers in the exchange of goods and services. It is commonly measured as the sum of consumer and producer surplus (SEEA-EA Glossary)." An economic welfare analysis allows for justification of the overall investment (or similar investments in future, based on this project), enables prioritisation of investment if selecting between multiple projects, and also facilitates the ability for different parties to co-fund investments based on the value of the ecosystem service changes that such a project produces. Here, we follow the SEEA-EA guidelines Chapters 8-11 to focus on measures of exchange value, but in accordance with Chapter 12 of the SEEA-EA guidelines, also include measures of welfare values where appropriate.

This guide provides a summary of different economic valuation methods that support estimation of exchange values for accounts:

- Figure 2.2 provides an overview of the relevant methods available to estimate exchange values for the different ecosystem services relevant to coastal wetlands and blue carbon restoration projects.
- Methods used to estimate welfare values are also captured in Figure 2.2 to assist in identifying where equivalent methods can be used to estimate both value measures.

- Approaches to estimate exchange values for different ecosystem services are captured throughout the subsections that follow.

In alignment with the SEEA-EA guideline, when compiling accounts, exchange valuation must be prioritised. **If monetary valuation of welfare values is included in an account** (to inform a decision-making process, for example), it must be explicitly stated and justified as to not cause confusion. Please refer to Chapter 12 of the UN SEEA-EA framework for further discussion and guidance on incorporating monetary valuation of welfare values into SEEA-EA accounts through the preparation of bridging tables[22]. Further discussion on application of welfare values in frameworks other than SEEA-EA can be found in **Appendix 2**).

Measurement and valuation at the site level

To estimate the benefits from a restoration project, at a minimum there is a need for an estimate of the level of activity or biophysical changes associated with the site, for example, visitation rates, or commercial production levels. This in many instances will require primary data collection. To then identify the monetary values associated with each 'unit' of activity (e.g. for each tonne of commercial fish extracted, or for each recreational trip made to a site) or other unit changes in ecosystem service outputs, requires further data. This would typically be acquired through primary data collection using surveys, and often these surveys are detailed and hence expensive to implement. For example, they may include surveys of firm/individual level outputs/costs, or of representative samples of recreational users, or the general public when estimating values associated with cultural services.

Such primary surveys to identify activity levels associated with ecosystem restoration may be technically feasible for sites with a sufficient stakeholder base, but could be relatively expensive to implement for small sites. Where primary data is not feasible to collect, use of average activity levels and average values for similar types of ecosystem services in similar locations could offer a more accessible alternative for compiling economic information. A more detailed discussion of extrapolating values through 'benefit transfer' (the process by which values from studies of different services may be adjusted and applied to the issue of interest) is provided in Section 12.2. In some cases, where primary or secondary data is not available to estimate the monetary benefits of ecosystem services directly, it is possible to instead refer to the costs avoided, for example through damage mitigation (also discussed in Section 12.2).

Identification of the monetary values enables ecosystem service accounts to be prepared. Where methods used to estimate the monetary values enable calculation of welfare values in addition to exchange values, and these are recorded in bridging tables, the information can also allow for total economic valuation or integrated economic assessment to be undertaken (discussed further in **Appendix 2**). An extension is to consider other economic indicators by using regional input output models to estimate regional multipliers for site specific expenditures; that is, direct expenditure in the region leads to additional expenditure by the recipients of that initial expenditure, leading to the multiplier effect. Regional multipliers are available in the literature and could be applied to give minimum (direct) and maximum (via multiplier) effects on the regional economy. For example, producing 1kg of prawns leads to approximately 5 times their initial value to the broader economy[24]. Multipliers are discussed further in Section 12.2.

A key aspect of measures of economic benefits (as defined here) is one of additionality. Identifying what is 'additional' means to identify the increase in activity or biophysical quantity associated with the restoration, relative to the activity and biophysical quantities that would be present without the restoration. It is important to consider substitution here too. Substitution occurs when there are multiple sites (or assets) within a proximity that are providing equivalent types of ecosystem services, and the consumers of those services might shift from the site they had been using to the restored site. In cases where there is substitution of economic activity from alternative sites as result

[22] United Nations, et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

[23] Voyer, M., et al. (2016). Social and Economic Evaluation of NSW Coastal Professional Wild-Catch Fisheries: Valuing Coastal Fisheries, University of Technology, Sydney, pg. 208. Report to Australian Fisheries Research and Development Corporation on Project 2014/301.

	Exchange values		Welfare values	
	Preferred methods	**Proxy methods**	**Preferred methods**	**Proxy methods**
First nations values			• Revealed preference • Stated preference	• Benefit transfer
Carbon sequestration	• Australian Carbon Credit Units • Reverse auction	• Certificate prices of Clean Development Mechanism • Averting behaviour • Avoided damage cost	• Social cost of carbon	• Stated preference
Water purification	• Expense for maintaining water quality • Expense for chemical removal	• Replacement cost • Avoided damage cost • Benefit transfer	• Revealed preference • Stated preference	• Benefit transfer
Coastal protection	• Defensive expenditure (e.g. construction cost of seawalls)	• Replacement cost • Avoided damage cost • Benefit transfer	• Revealed preference • Stated preference	• Avoided cost • Replacement cost • Benefit transfer
Fisheries biomass	• Market price		• Profit	
Recreational services	• Total trip expenditure	• Travel cost • Simulated exchange value • Benefit transfer	• Revealed preference • Stated preference	• Benefit transfer
Existence value			• Stated preference	• Benefit transfer

Figure 2.2: Valuation methods for different ecosystem services. Exchange valuation methods based on the official methods recommended by NCAVES and MAIA (2022)[24] Table 6.4.

[24] NCAVES and MAIA (2022). Monetary valuation of ecosystem services and ecosystem assets for ecosystem accounting: Interim Version 1st edition. United Nations Department of Economic and Social Affairs, Statistics Division, New York.

of the restoration, one cannot claim the full level of economic activity (production/jobs etc.) as a direct benefit associated with the change in value of the service flows at the restored site as there will have been reductions in activity from elsewhere. For example, aggregate combined expenditure by recreational fishers may not change as they shift site choice resulting from restoration: indeed, it may be reduced if the restoration activity means that (collectively) fishers do not have to travel as far.

The issue of attribution is important when revealed preference approaches such as hedonic price models (discussed further in Section 12.2), which involve breaking down the characteristics of a good and attributing value to each characteristic, are employed, as otherwise there is a risk of double counting benefits that have been evaluated elsewhere.

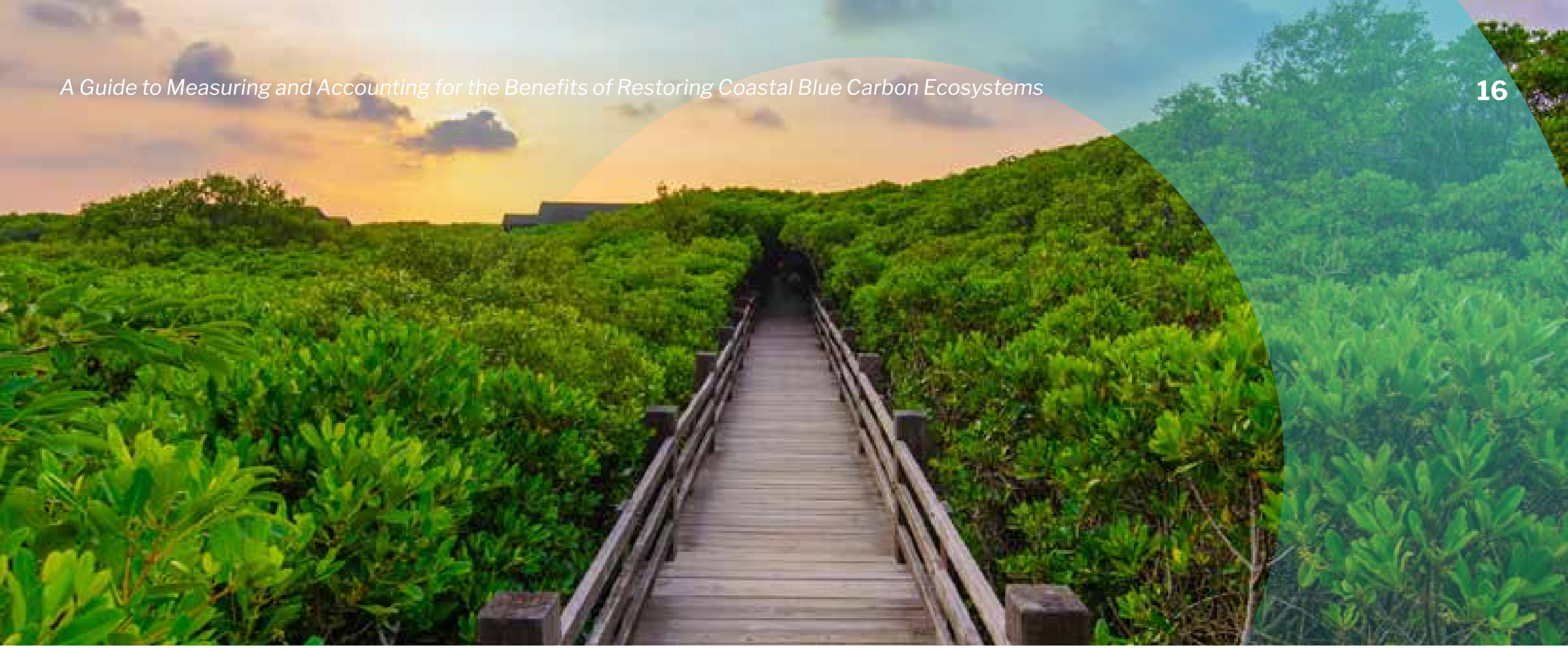
Ecosystem services explored in this Guide

The SEEA - EA provides a reference list and description of ecosystems services, the relevant components of which are given in Table 2.1. The following sections provide detailed discussion of the main ecosystem services that may change due to a restoration of a coastal blue carbon ecosystem:

- Carbon sequestration and emissions
- Water purification services
- Coastal protection: erosion, storm mitigation and flood control services.
- Fish production: nursery habitat services
- Fish production: biomass provisioning services
- Cultural services: recreation and non-use values
- Cultural services: First Nations values

Table 2.1: List of selected Ecosystem Services considered in the Guide and their definitions from the SEEA–EA guidelines.

Ecosystem Service		Description
Provisioning services		
Biomass provisioning services	Wild fish and other natural aquatic biomass provisioning services	Wild fish and other natural aquatic biomass provisioning services are the ecosystem contributions to the growth of fish and other aquatic biomass that are captured in uncultivated production contexts by economic units for various uses, primarily food production. **This is a final ecosystem service**.
Regulating and maintenance services		
Global climate regulation services		Global climate regulation services are the ecosystem contributions to reducing concentrations of Green House Gases in the atmosphere through the removal (sequestration) of carbon from the atmosphere and the retention (storage) of carbon in ecosystems. These services support the regulation of the chemical composition of the atmosphere and oceans. **This is a final ecosystem service**.
Water purification services (water quality regulation)	Retention and breakdown of nutrients	Water purification services are the ecosystem contributions to the restoration and maintenance of the chemical condition of surface water and groundwater bodies through the breakdown or removal of nutrients and other pollutants by ecosystem components that mitigate the harmful effects of the pollutants on human use or health. **This may be recorded as a final or intermediate ecosystem service**.
Soil and sediment retention services	Soil erosion control services	Soil erosion control services are the ecosystem contributions, particularly the stabilising effects of vegetation, that reduce the loss of soil (and sediment) and support use of the environment (e.g. agricultural activity, water supply). **This is may be recorded as a final or intermediate service**.
Flood control services	Coastal protection services	Coastal protection services are the ecosystem contributions of linear elements in the seascape, for instance coral reefs, sand banks, dunes or mangrove ecosystems along the shore, in protecting the shore and thus mitigating the impacts of tidal surges or storms on local communities. **This is a final ecosystem service**.
	River flood mitigation services	River flood mitigation services are the ecosystem contributions of riparian vegetation which provides structure and a physical barrier to high water levels and thus mitigates the impacts of floods on local communities. River flood mitigation services will be supplied together with peak flow mitigation services in providing the benefit of flood protection. **This is a final ecosystem service**.
Nursery population and habitat maintenance services		Nursery population and habitat maintenance services are the ecosystem contributions necessary for sustaining populations of species that economic units ultimately use or enjoy either through the maintenance of habitats (e.g. for nurseries or migration) or the protection of natural gene pools. **This service is an intermediate service** and may input to a number of different final ecosystem services including biomass provision and recreation-related services.
Cultural services		
Recreation-related services		Recreation-related services are the ecosystem contributions, in particular through the biophysical characteristics and qualities of ecosystems, that enable people to use and enjoy the environment through direct, in-situ, physical and experiential interactions with the environment. This includes services to both locals and non-locals (i.e., visitors, including tourists). Recreation-related services may also be supplied to those undertaking recreational fishing and hunting. **This is a final ecosystem service**.
Spiritual, artistic and symbolic services		Spiritual artistic and symbolic services are the ecosystem contributions, in particular through the biophysical characteristics and qualities of ecosystems, that are recognised by people for their cultural, historical, aesthetic, sacred or religious significance. These services may underpin people’s cultural identity and may inspire people to express themselves through various artistic media. **This is a final ecosystem service**.
Flows related to non-use values		
Ecosystem and species appreciation		Ecosystem and species appreciation concerns the wellbeing that people derive from the existence and preservation of the environment for current and future generations, irrespective of any direct or indirect use.



3. Process of implementation

The purpose of this guide is to assist those seeking to measure the benefits of a coastal blue carbon ecosystem restoration project to rigorously identify and measure outcomes for climate, ecosystems, and people, and report outputs including using SEEA-EA aligned tables. As such, for the purposes of this document it is assumed that restoration projects have been defined separately from this process, and this guide focuses on measuring and presenting the outcomes of restoration projects rather than conducting restoration activities and planning.

The guide proposes the following steps to identify and measure the environmental and economic outcomes of a coastal blue carbon restoration project. Some components (e.g. account design and compilation) are more relevant to preparing SEEA-EA aligned tables as an end product, but provide useful information for anyone seeking to assess and report on the outcomes of a restoration project:

- **Project scoping and framing**: to consider the geographic and temporal scope of interest to the analysis, identification and description of ecosystem types and anticipated benefits of restoration, the restoration intervention itself, and desired outputs. Budget availability is also important to consider here.
- **Data collection**: this involves a detailed consideration of the restoration activities undertaken, anticipated outcomes of the restoration project and those of most interest including within and outside of the project boundaries.
- **EEA accounts - Identify and assemble data**: across all relevant aspects of the accounts, collate data and transform into coherent format.
- **EEA Account compilation**: develop fully populated accounts representing the environmental and economic conditions of the project area before project implementation.
- **Repeat data collection and continue building accounts**: to measure changes in the key variables of interest over time and demonstrate effects of restoration.
- **Report results**: using repeated tables over time, the results of the restoration project can be articulated in an internationally recognised format.

As will be discussed in subsequent sections, the process of account implementation is iterative, with many steps and possible feedback loops. The first design of the accounts will represent the “ideal” suite of accounts and the ecosystem assets and services to present. It is rarely possible to populate such accounts completely and it is often the case that the accounts will be re-designed based on the data available for use in the accounts, unique attributes of the ecosystem, and the ability of this data to be harmonised and compliant with the SEEA-EA.

More detail on these steps is provided in Figure 3.1, along with other information relevant to a proponent aiming to identify and record the impacts of a restoration project.

3.1 Project scoping

As identified above, it is assumed in this document that the identification and scoping of the restoration project itself has been done outside of the process for measurement and account development. However, preparing an assessment starts with understanding the restoration project including:

- **Spatial coverage**: considering the physical area of the suite of anticipated outcomes from the intervention. This may differ from the boundaries of the intervention itself, for example if a fishery adjacent to the intervention receives an increased population of key species.
- **Temporal coverage**: the duration of time that project reporting is expected to cover.
- **Defining and describing ecosystems**: define and describe ecosystems using local knowledge and if developing EEA accounts, aligning with the recently released Global Ecosystem Typology[25].
- **Ecosystem extent and condition**: considering the ecological communities within the project area and how their condition might change due to the restoration intervention (and appropriate data points to measure these).
- **Ecosystem service scoping**: considering the types of ecosystem services produced within the project area, and their beneficiaries, particularly those that might be changed by the restoration intervention.
- **Effects of intervention**: knowing the method of restoration intervention, identifying the effects on extent and condition of ecosystem stocks, and anticipated benefits of restoration. Priority effects which determine most of the changes in ecosystem function should be identified as the most important to measure. This refers to ecosystem services in particular (flows).
- **Stakeholder mapping**: identifying stakeholders to consult to best understand the effects of the project, collect relevant data, and provide a social license for restoration approaches.

Developing a conceptual model of the ecosystem that depicts relationships between the identified features, and key processes and threats is a critical step in project scoping. A conceptual model makes it easier to identify the drivers of change and value in an ecosystem, the best ways to monitor the ecosystem, and knowledge gaps that need to be addressed. Conceptual models are effective tools for communicating restoration and accounting projects to stakeholders in an easily digestible way. Conceptual model development is covered in detail in Section 6: Ecosystem condition.

The elements above are discussed in detail below and in subsequent sections but should be scoped out at the start of the project. One of the tools most appropriate to determine all these aspects is a conceptual model of the ecosystem, and how it interacts with society. Conceptual models are helpful to identify gaps in knowledge and are also helpful to stakeholders familiar with the area who may advise reframing or adding/removing aspects that have been identified within the conceptual model. See Section 6 for further detail and examples on creating conceptual models.

[25] Keith, D. A., et al. (2022). A function-based typology for Earth’s ecosystems. Nature, 610(7932), 513–518. https://doi.org/10.1038/s41586-022-05318-4

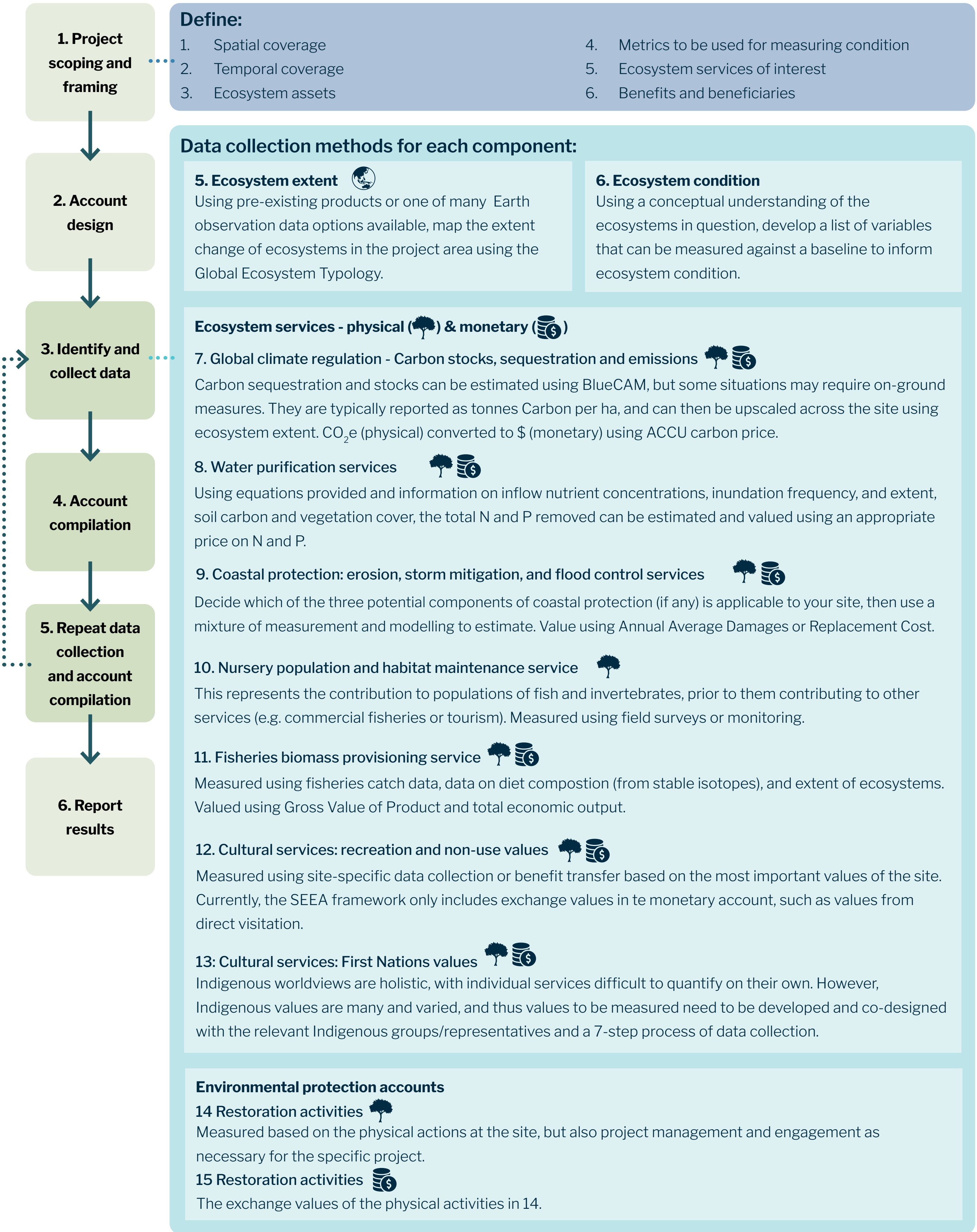
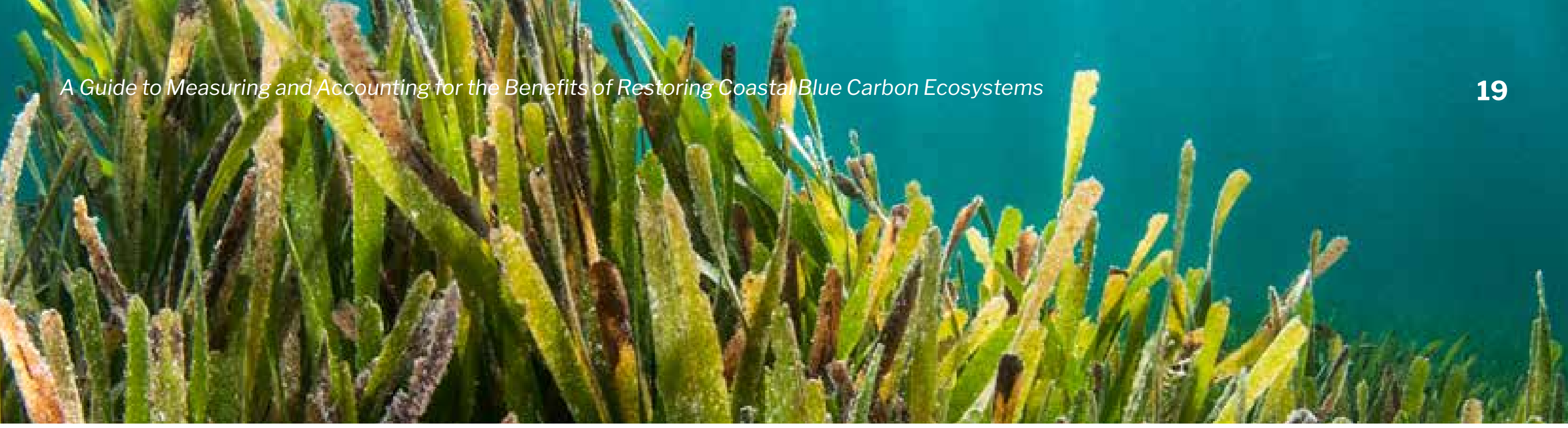


Figure 3.1: Steps in compiling a benefit assessment of restoring blue carbon ecosystems including using EEA. Accounts where physical services are quantified are denoted by the tree icon and accounts where monetary services are quantified are denoted by the money icon.



3.2 Data needs

Data collection and management is central to any project aiming to successfully conduct environmental economic accounting. Once the anticipated restoration benefits or interest have been identified, sourcing the types of data required should be a key objective. In many cases, some data will already exist and be available for sourcing, while other data may require primary data collection. These data also typically range in quality depending on the site and previous research that may or may not have occurred in the area.

The methodologies described in Section 5: Ecosystem extent, include discussions on data needs as well as existing data sources that can be drawn upon to measure different restoration outcomes. Users of the guide are also encouraged to review the example EEA account tables at the end of this document, and the completed tables in the associated case studies, to consider how the restoration benefits might be presented and the associated data needs. Understanding the types of data required to populate account tables will be helpful to users to identify their data requirements.

Basic EEA account structures

When preparing accounts, the three main types of physical ecosystem accounts are for ecosystem extent and ecosystem condition, and ecosystem supply and use tables. Examples of what these tables look like are provided in Section 6. An ecosystem extent change matrix is also commonly prepared as a full accounting asset, which allocates the changes in extent to natural or anthropogenic factors, using available data. Monetary accounts for ecosystem assets and ecosystem services are also part of the SEEA-EA. The full accounts are presented in the SEEA-EA[26].

3.3 EEA account compilation

The compilation of a set of EEA accounts is the final step in the first iteration of an accounting process for projects wishing to present their restoration benefits data in this format. In this, basic accounting identities are checked. Specifically, that supply equals the use of ecosystem services and the changes between opening and closing stocks in the asset accounts balance. Where these reflect a physical increase or decrease in stocks due to flows of ecosystem services, which is the case for carbon sequestration and carbon storage for example, then the asset account and the supply and use tables must be aligned.

Assigning the use of ecosystem services and the reasons for change in the ecosystem extent and condition accounts is often a challenging task. In some cases, it can require seeking additional information and is part of the iterative process of account production.

Each account produced should be accompanied by suitable descriptions of the data sources and methods used in their compilation.

3.4 Repeat data collection and EEA account compilation

To assess the changes in ecosystem accounts due to restoration, data need to be collected before and after restoration actions occur. Data may need to continue to be compiled for many years after restoration actions have been undertaken due to the time taken for changes in abiotic and biotic components of the ecosystem, and then flow on effects to services.

Systematic annual data is needed for account compilation, but other accounting periods can

[26] United Nations, et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

be defined depending on the goals of the project (for example, 5-year intervals). Standard statistical processes should be used to obtain samples representative of the total accounting area for all the ecosystem assets and services included in the accounts. Over time the data sources and methods used in the production of accounts may improve[27]. Ecosystem restoration projects can take quite some time to implement (potentially several years), and then the effects produced from the projects may take several years to observe. The construction of other types of accounts, such as for ecosystem potential (e.g. the maximum area that could be covered by a habitat) could provide information on expected future benefits. In the case of some key impacts associated with certification (such as carbon sequestration), ongoing annual data collection may be necessary.

3.5 Report results

The development of annual accounts for the project area over time will allow for the environmental and economic impacts of a restoration project to be clearly articulated. The accounts, along with a transparent articulation of data inputs and any assumptions used, will demonstrate the impact of the project and can be drawn upon for the certification of any project outputs, such as carbon, coastal protection, or biodiversity credits[28].

Section 16: EEA outputs and presentation, presents a set of reporting tables aligned with the SEEA-EA that were prepared for case studies undertaken for this project. These could be drawn upon by those seeking to prepare account outputs for their restoration project.

3.6 Critical considerations

Other considerations are relevant to those wishing to use accounting methods to demonstrate the impacts of a restoration project:

Skills and expertise

The implementation of a restoration outcomes measurement; and if desired, a SEEA accounting process for a blue carbon ecosystem restoration project; requires technical expertise across a range of disciplines, as well as project management skills. Contributing input will be required from the following key technical skillsets:

- **Scientists and ecologists**: responsible for identifying, categorising, and measuring the biophysical changes, and the ecosystem and climate benefits of the restoration project.
- **Economists**: responsible for identifying, categorising, and measuring the values, not necessarily in monetary terms, of the ecosystem assets and ecosystem services from the restoration project.
- **Social scientists**: responsible for identifying, categorising, and measuring the cultural benefits experienced from restoring coastal blue carbon ecosystems.
- **EEA Accountants**: responsible for compliance with EEA accounting concepts, definitions and conventions and the design of the accounts for ecosystem assets and ecosystem services of the project area.
- **Project managers**: additional to the above technical experts, as with any complex project involving technical experts and stakeholder engagement, project management expertise will assist in the implementation of the project and delivery of outputs.

More complex accounts across wider areas will need greater input from experts than for smaller projects.

[27] Vardon, M., et al. (2018). How the System of Environmental-Economic Accounting can improve environmental information systems and data quality for decision making. Environmental Science & Policy, 89, 83-92. https://doi.org/10.1016/j.envsci.2018.07.007

[28] Users should of course confirm the certification requirements of outputs with the relevant certifying body.

As more accounting projects on blue carbon ecosystems occur, it is expected that the time and input required from each of these experts will decrease as more information will be available to draw on from previous projects in similar systems. The input from these experts over the duration of a project will also decrease over time, as the initial account creation requires most of the effort, with updates to the accounts being more streamlined.

Stakeholder engagement

Engagement with stakeholders and their representatives in the project area is an integral part of any assessment process and should be considered within the project design. Stakeholder engagement will assist in:

- Identifying and scoping the key ecosystem-related activities that take place within the project area.
- Identifying key representatives who can speak to Indigenous values associated with the project area and assist in establishing these values.
- Identifying key data sets that may be held by those with expertise in the local area.
- Establishing scale and scope of some key values, particularly cultural and social ones.
- On-going production of the accounts (including redesign and refinement of accounts to better suit stakeholder needs).
- Obtaining a social license to undertake restoration projects.

Stakeholder engagement could involve workshops with different stakeholder groups, individual interviews in person or remotely as is appropriate.

4. Summary of key components

This section provides brief summaries of the main components of an impact assessment of a coastal blue carbon restoration project. Each component is described, along with key data sources to be used, and a brief discussion of methodologies recommended for measuring them. Each subsection is elaborated on in some detail, in the sections that follow.

4.1 Extent accounts

Ecosystem extent refers to the areal extent of different ecosystems present within an area of interest. A key measure of the success of restoration activities in blue carbon ecosystems is its change in ecosystem extent, typically measured as change from the pre-restoration extent to the post-restoration extent (as shown in Figure 4.1). An increase in areal extent of blue carbon ecosystems after restoration activities implies a good outcome where this is the goal of the project. Indicators of restoration success should target the objectives of restoration, which align with value propositions of the restoration activities (e.g. increase in saltmarsh as habitat for water birds).

Measuring coastal ecosystem extent change can be achieved using remote sensing to define vegetation community boundaries, produce maps of vegetation community distributions and quantify extents. A key component of using Earth observation data and remote sensing techniques is validation, which refers to assessing the accuracy or uncertainty of remote sensing products; this is often undertaken by comparison with analytical reference data (such as corresponding ground and field measurements or using experts to verify). Remote sensing approaches can be cost effective, reproducible, and standardised, and are effective for measuring coastal ecosystem extent and changes in extent over time. They can also provide information about biophysical and structural characteristics of coastal vegetation communities that can be useful for quantifying changes in condition (discussed further below in Section 4.2.

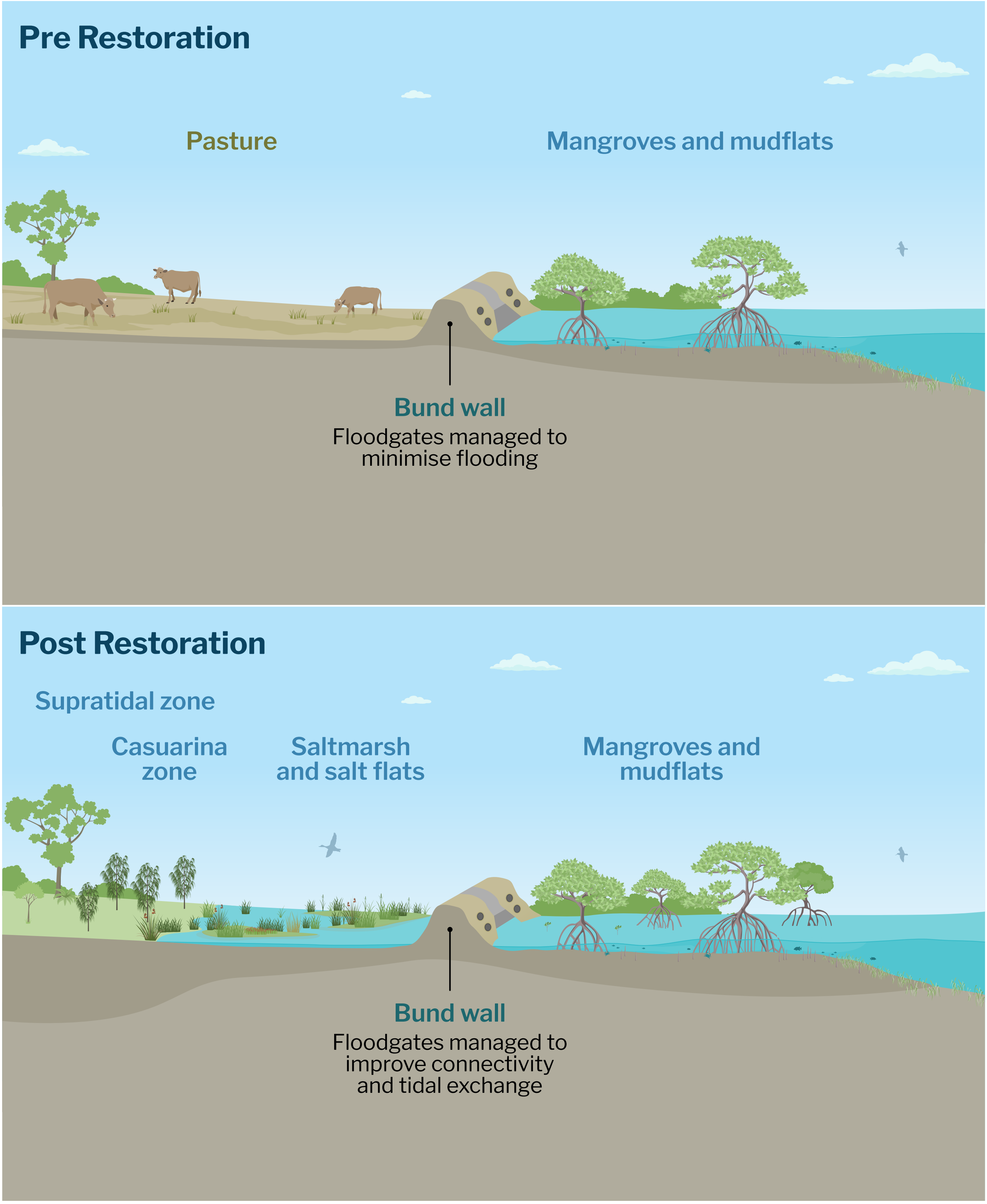


Figure 4.1: Ecosystem extent change following restoration.

Methods

Conceptually, measuring ecosystem extent is relatively straight forward. First, ecosystem types and level of detail of differentiation need to be defined. In alignment with the SEEA-EA guidelines, we recommend using the Global Ecosystem Typology[29] to define and describe ecosystems[30]. For a simple coastal wetland restoration project this may be mangrove, saltmarsh, and grassland pastures. Secondly, these ecosystem types are delineated spatially, whereby distinct ecosystems are mapped. Finally, the delineated areas of post-restoration are subtracted from pre-restoration ecosystem extents and net change in each ecosystem extent can be quantified.

Measuring changes in ecosystem extent using Earth observation data and remote sensing technologies requires skills in spatial science. The level of experience and expertise will vary with the availability of data, and level of detail required to detect changes in ecosystem extent. Where national products are available and appropriate for project-level ecosystem accounts, moderate skills in geographic information systems (GIS) will be required. Most national products will be available in formats suitable for use in a GIS, typically raster format, and can be extracted from data portals. Digital Earth Australia (DEA) provides access to national mapping products, such as DEA mangroves, and it is anticipated that similar data products will be available for saltmarsh, supratidal forests, and seagrass in the future. Using national products presents a lower cost method that will provide sufficient level of detail for large restoration sites where considerable ecosystem extent change is anticipated.

Where national products are not available or not suitable for identifying project-level extent and change to extent over time, moderate to high GIS and remote sensing expertise will be required to generate suitable mapping products. Depending on GIS expertise available, multiple approaches can be taken to provide sufficient rigor. This approach provides capacity to undertake a more detailed approach (e.g. higher resolution data) potentially leading to greater confidence in ecosystem extent measures. A detailed approach presents a higher cost method than using national datasets, however, may be required due to project level extents that may be at a smaller scale or where ecosystem extent change is more complex due to impact on the ecosystem or restoration intervention.

Data sources

There are a variety of Earth observation data and remote sensing technologies available for quantifying changes in ecosystem extent. Active sensors, such as Lidar and radar, provide useful information about the structure and distribution of coastal vegetation communities, whilst passive sensors provide spectral data that can be used to derive ecosystem extent. These sensors can be used from space-borne, airborne, and remotely piloted aircraft; the aircraft that the sensor is affixed to and its height above the Earth will modify the resolution, precision, and accuracy of remotely sensed data. Similarly, the resolution of the sensor can modify the resolution, precision, and accuracy of remotely sensed data.

Selection of Earth observation data and remote sensing technologies should be based on suitability and availability for mapping changes in ecosystem extent. Measuring the extent of each coastal vegetation community is important as the ecosystem services they provide vary, as do the restoration activities undertaken due to varying environmental settings and pre-impact and pre-restoration condition. Coastal vegetation communities can be differentiated based on spectral, structural and elevation range characteristics.

Mapping ecosystem extent can be achieved at multiple spatial and temporal scales. For project level environmental economic accounts, extent calculations are influenced by the resolution of Earth observation data, mapping approaches and overall accuracy of vegetation community boundaries. However, production of highly accurate maps should be balanced against the costs and expertise of production to provide sufficient rigour.

[29] Keith, D. A., Ferrer-Paris, J. R., Nicholson, E., & Kingsford, R. (2020). IUCN Global Ecosystem Typology 2.0: descriptive profiles for biomes and ecosystem functional groups (D. A. Keith, J. R. Ferrer-Paris, E. Nicholson, & R. T. Kingsford (Eds.)). IUCN, International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2020.13.en

[30] Keith, D.A., et al. (2020). The IUCN Global Ecosystem Typology 2.0: Descriptive profiles for biomes and ecosystem functional groups. Gland, Switzerland: IUCN. https://portals.iucn.org/library/sites/library/files/documents/2020-037-En.pdf

Using existing mapping products, where available, will minimise costs and expertise requirements. Approaches that can be implemented using readily available datasets that are available at a national scale will ensure standardisation of accuracy and precision. Several existing products can be used to map ecosystem extent that are publicly available. At the project-level, these datasets may not be useful to identify extent change for accounts due to the resolution of imagery, which can limit the capacity to detect changes in extent (e.g. Landsat imagery resolution of ~ 25-30 m); and the temporal availability of datasets, which can limit the capacity to measure ecosystem extent pre-restoration (e.g. Landsat imagery suitable for monitoring extent available from 1987 onwards).

Where existing products do not meet the needs of the project, additional data and analyses will be required. Suitable protocols and methods should be followed when using alternative data sources to ensure both suitability for the project and comparability to other project-level restoration activities.

For a detailed discussion of ecosystem extent measurement, see Section 5.2.

4.2 Condition accounts

Measuring improvements in the condition of ecosystems due to restoration actions is a key component in measuring a restoration project's success. Condition of an ecosystem includes environmental aspects, such as salinity and nutrient levels, as well as measures of the living parts of the ecosystem, including plant structure and biodiversity.

While extent accounts measure change in the amount or area of the ecosystem, condition accounts measure the changes in the multiple aspects of the ecosystem, relative to a 'natural state' for that ecosystem type (Figure 4.2, in an Australian context (as with other colonised lands), 'natural state' would be the pre-colonisation state). For instance, are the habitat-forming species sparse or dense, tall or short, do they provide enough of the right habitat for species? The reference level may be quite different to the restoration goal, recognising that it may not be possible to restore an ecosystem back to its 'natural state' due to the influence of dynamic processes such as sea-level rise or, changes in rainfall, or declines and extinction of species.

Condition is also more than just metrics of the habitat forming species (e.g. mangroves). It is the condition of the water, soils, and air the plants and animals live in, and the biodiversity and functioning (e.g. plant productivity) of the species that inhabit these ecosystems. Measures of ecosystem condition do not need to be linked to ecosystem services, although this can be done separately as part of those accounts.

Methods

Figure 4.3 sets out the steps to take to assess condition. Given that there are so many potential condition variables to measure, it is important to choose these based on the knowledge of threats to the site, the current and potential future values of the restoration site. Practitioners and experts in condition assessment should be engaged to do this. The first two steps set goals and area (common to all accounts). To plan your condition assessment, it's recommended to develop a simple diagram (conceptual model) of the site and ecosystems. From there, you can choose the condition variables and indicators and align these with the Ecosystem Condition Typology (outlined further in Table 2.1 and Table 5.1). You will also need to decide what a 'natural state' is for the site (what is your measure of "good condition" for the coastal ecosystem), by comparing the restoration site to nearby reference sites, or choosing the reference level using a range of other options detailed further below. Decide on the sampling frequency, regime and standardised approach which depends on the condition variables you have chosen and goals and timeline of the project. Then you can begin.

Once the condition account has been designed, you will need to find the appropriate methods to measure your variables of interest. The guide outlines various methods and approaches and categorised them as either high or low cost, and our recommendation for monitoring of restoration sites. However, this is based on the science as it currently stands, and there are a number of novel, low cost monitoring methods in development which may be able to be implemented in the future.

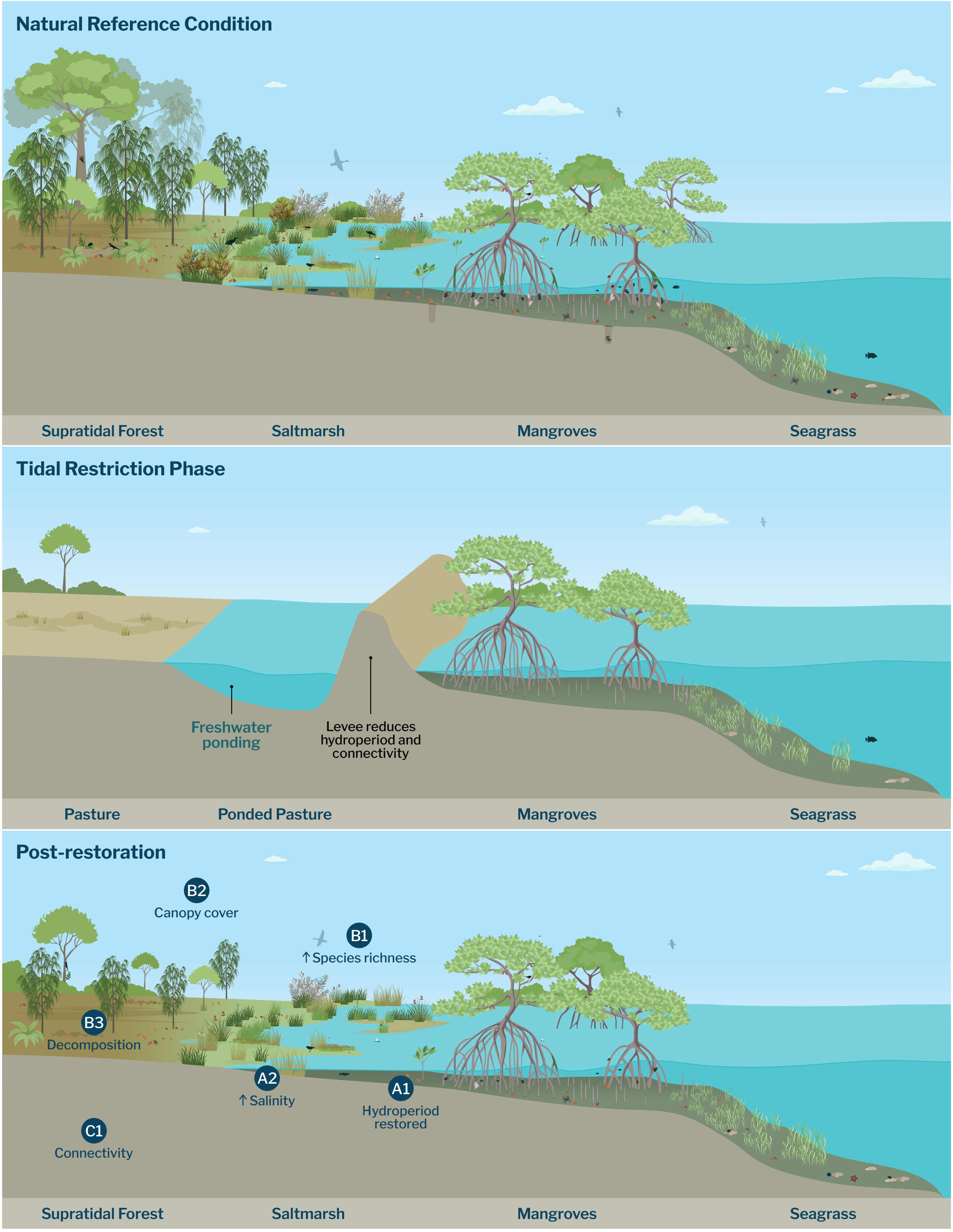


Figure 4.2: **Measures of ecosystem condition include measures of the water, soil and air, in addition to measures of the habitat forming species, biodiversity, and connectivity between them all (indicators within the SEEA Ecosystem Condition Typology of Physical state [A1], Chemical state [A2], Compositional state (biodiversity measures) [B1], Structural state [B2], Functional state [B3] and Land/Sea-scape characteristics [C1]).** Upper left-hand side indicates pre-restoration condition, right-hand side indicates natural reference/goal condition, lower panel indicates post-restoration condition. Codes refer to examples of variable types given in Table 2.1 and Table 5.1.

- Step 1: Identify goals of restoration activity
- Step 2: Establish restoration areas of interest
- Step 3: Develop conceptual model for the area of interest
- Step 4: Identify appropriate condition metrics
- Step 5: Choose appropriate approach to estimate natural reference condition
- Step 6: Define the sampling regime
- Step 7: Determine approach for standardising condition metrics
- Step 8: Undertake condition assessment and standardisation for each condition metric

Figure 4.3: Flowchart to assess condition.

Data sources

Data sources for condition can include spatial and field derived sources, and collections as part of planned monitoring. Data availability, resourcing available, context and purpose of the restoration project will inform which sources are used. Multiple spatial data sources are now freely available and can be utilised by qualified spatial analysts. This guide provides suggested data sources that can be utilised including databases and repositories.

For a full discussion of ecosystem condition, please see Section 6: Ecosystem condition.

4.3 Climate regulation accounts

Coastal wetlands are recognised for the disproportionate role they play in global carbon cycling, relative to their spatial extent, based upon three main factors:

1. Coastal wetlands are productive ecosystems, meaning they produce a lot of plant biomass each year, drawing large amounts of carbon dioxide from the atmosphere, which is stored aboveground and belowground in plant roots, rhizomes and in soils.

2. Frequent inundation by tidal waters reduces exposure of soils to oxygen, slowing the decomposition of organic matter, and leading to long-term storage of carbon. Meanwhile, saline waters minimise the production of methane, which is a powerful greenhouse gas and significant component of emissions from some freshwater settings.

3. 3. There are substantial opportunities for restoring coastal wetlands. Such actions can have the combined benefit of (1) reducing existing greenhouse gas emissions from degraded coastal landscapes; and (2) removing additional carbon dioxide from the atmosphere by the newly restored/created habitats where carbon is stored in plant biomass and soils (Figure 4.4).

Restoration activities in the coastal zone can influence the coastal carbon cycle in multiple ways. For example, interventions that modify the frequency, duration and/or seasonality of inundation may influence pre-existing (baseline) emissions of carbon dioxide, methane, and nitrous oxide. Changes to inundation regimes may also modify vegetation composition and productivity, rates of carbon decomposition, and the sedimentation or erosion of carbon-rich materials. Changes to

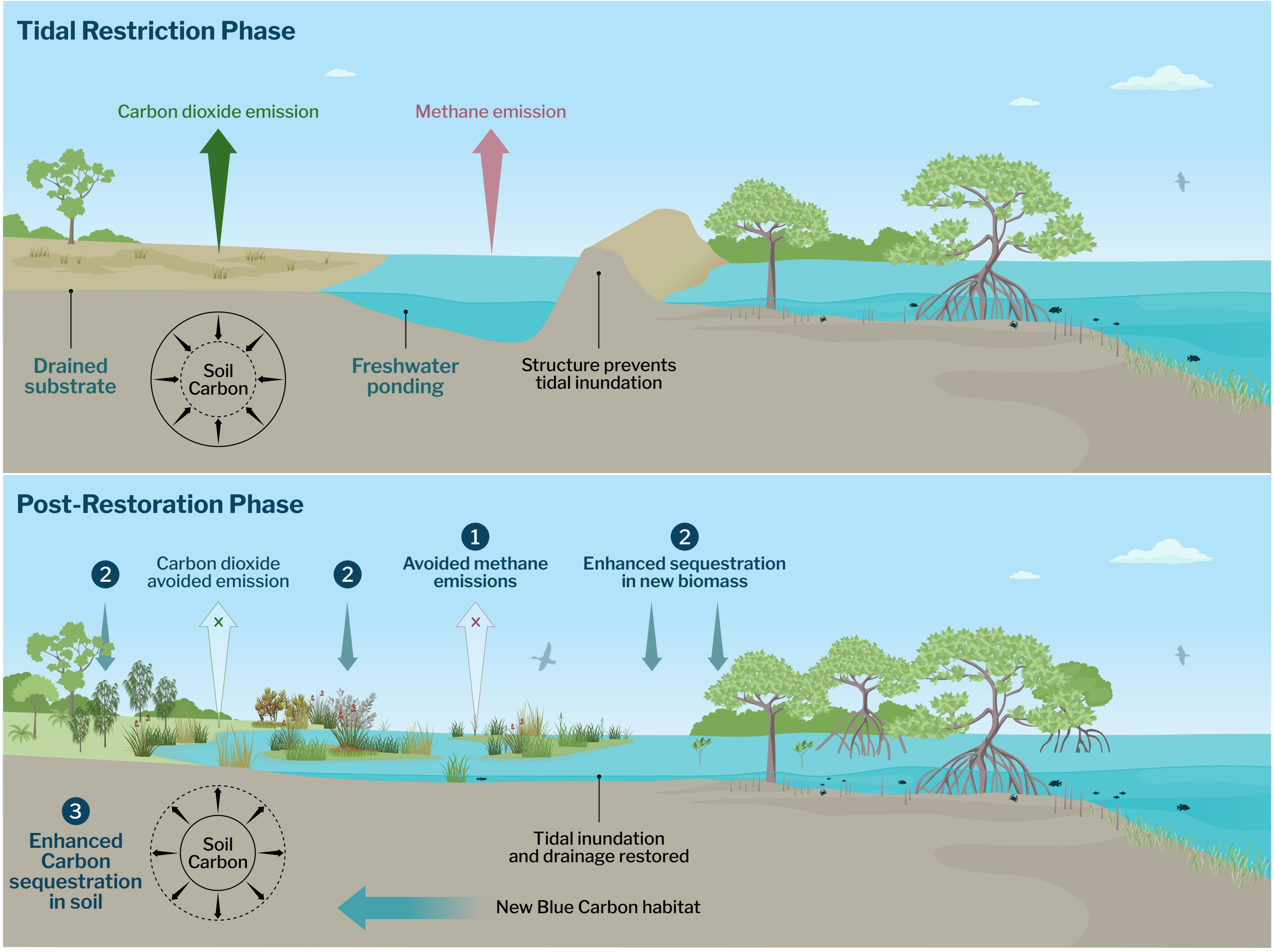


Figure 4.4: Carbon services conceptual figure.

water or soil chemistry may similarly influence vegetation composition and productivity, and/or the rates at which organic matter decomposes. Such changes may result in either positive or negative outcomes for greenhouse gas emissions, carbon sequestration and carbon stocks, with the direction and magnitude of outcomes dependent upon the land use transitions involved, extent, location, and timeframe of restoration actions.

Methods

This guide details an integrated approach for quantifying three related, though distinct, accounts associated with global climate regulation: (1) carbon abatement; (2) carbon sequestration; and (3) carbon stocks/storage. The carbon abatement account integrates estimates of greenhouse gas emissions and carbon sequestration through the life of a restoration project (i.e. years to decades) to determine the net outcomes of carbon abatement of tidal restoration actions at this site. This account includes both physical and financial accounts.

In contrast, the carbon stock/storage account provides snapshots of the amount of carbon stored in aboveground biomass and soil carbon (to 1 m depth) pools within the study area, estimated at two time points: (1) a pre-restoration time point; and (2) a post-restoration time point. No financial account has been estimated for carbon stocks as this would represent double-counting of values which are already considered in the carbon abatement account.

Project proponents can choose from two-tiers for estimating the physical accounts of carbon abatement, carbon sequestration, and carbon stocks/storage services, depending on their data and resource availability:

- **Nationally-consistent approach** – low cost approach utilising existing nationally-available datasets and a variation of the Commonwealth Government's Blue Carbon Accounting Model (BlueCAM) calculator; or
- **Detailed approach** – integration of setting-specific and high-resolution datasets with the BlueCAM calculator to provide more accurate estimates of carbon accounts.

This guide provides methods and templates for the compilation of carbon abatement, carbon sequestration and carbon stock accounts over the life of a restoration project following the BlueCAM accounting framework[31]. Additionally, guidance is provided for the compilation of carbon storage and carbon sequestration accounts only (i.e. excluding emissions and net carbon abatement) under the SEEA framework, which requires separate estimation of single year accounts at the beginning and end of a project accounting period.

Data sources

The Blue Carbon Accounting Model (BlueCAM)[32,33] calculator is a repository of nationally relevant datasets and accounting procedures and is therefore a central tool for generating the carbon accounts described in this guide. BlueCAM requires some project specific parameters such as tidal range, elevation and land type/ecosystem extent accounts which are detailed in other sections of this guide. Where available, site-specific datasets of carbon cycling parameters may also be sourced from direct measurement, the literature, online repositories, or reliable unpublished sources to complement BlueCAM default values and provide more accurate estimates of carbon accounts.

For a full discussion of climate regulation measurement, please see Section 7: Carbon stocks, sequestration & emissions.

4.4 Water purification services

Restoring coastal wetlands can improve water quality by reducing nutrients (nitrogen, N, phosphorus, P) and total suspended solids (TSS) from the water column. Nitrogen can be removed through denitrification, the process in which soil microorganisms convert nitrate (NO_3^-) to gaseous nitrogen (N_2, Figure 4.5). Trees can also remove nitrogen (primarily as ammonia, NH_4^+) and dissolved phosphorus (e.g. as phosphates PO_4^-) and store it in their wood. Additionally, sediment accretion can retain total suspended solids, particulate nitrogen, and phosphorus in the wetland. Finally, in cases where wetland restoration objectives include remediating acid sulphate soils, tidal flushing will restore natural acidity (pH) values in the water and sediment.

To quantify the values of the restoration of a coastal wetland for improving water quality, four main processes should be considered: 1) denitrification, 2) tree uptake, 3) sedimentation/sequestration, and 4) acidity regulation.

Methods

This section proposes how to determine the potential of water quality improvement through restoration for each process that contributes to reductions in nutrients and suspended sediments. This should be done in the following steps:

1. Establish wetland type and vegetation area.
2. Determine denitrification potential from nitrate concentration in the floodwater, frequency of inundation and vegetation cover.
3. Determine N and P uptake through tree uptake from the increase in woody biomass as tree growth, forest density and wood N and P concentrations.

[31] This Guide is not directly applicable to the preparation of projects under the ACCU Scheme's Tidal Restoration of Blue Carbon Ecosystems method, nor for the management and reporting requirements of approved projects under that method. Appropriate guidance is provided in the Tidal restoration method[34].

[32] Clean Energy Regulator (2022). The blue carbon accounting model (BlueCAM). Clean Energy Regulator. https://www.cleanenergyregulator.gov.au/DocumentAssets/Pages/The-blue-carbon-accounting-model-BlueCAM.aspx

[33] Clean Energy Regulator (2022). Tidal restoration of blue carbon ecosystems method. Clean Energy Regulator. https://www.cleanenergyregulator.gov.au/ERF/Choosing-a-project-type/Opportunities-for-the-land-sector/Vegetation-methods/tidal-restoration-of-blue-carbon-ecosystems-method

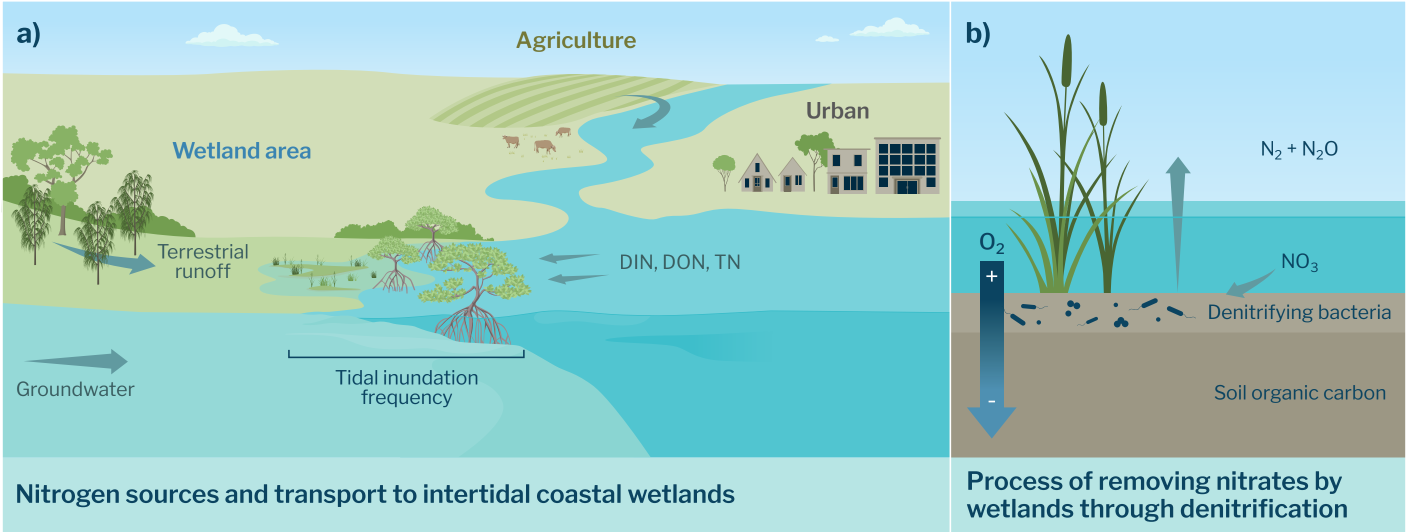


Figure 4.5: a) Nitrogen moving from terrestrial sources through rivers, surface runoff, and groundwater to coastal wetlands as total (TN), dissolved inorganic (DIN= NO_3-+ NH_4+) and organic nitrogen (DON), b) wetlands remove NO_3- through denitrification by microorganisms in waterlogged sediments with low oxygen.

4. Determine N and P retention through sedimentation in the wetland through estimations of sediment accretion and measurements of N and P in surface soil.
5. Determination of total suspended sediment retention through sedimentation in the wetland through information on turbidity (NTU) of water inundating the wetland.

Data sources

1. The first data set required is the extent of vegetation types before and after restoration. These can be obtained from the extent and condition accounts (Section 5 and Section 6).
2. The second data set is required to determine the water quality that the restored wetland can potentially treat. Multiple programs regularly monitor water quality; for instance, State-led long-term water quality monitoring programs include the following:
   - QLD: https://www.qld.gov.au/environment/water/quality/monitoring
   - NSW: https://www.environment.nsw.gov.au/topics/water/water-quality/monitoring-and-reporting
   - VIC: https://data.water.vic.gov.au/
   - WA: https://www.water.wa.gov.au/water-topics/waterways/threats-to-our-waterways/statewide-river-assessment
   - TAS: https://nre.tas.gov.au/water/water-monitoring-and-assessment/water-monitoring/surface-water-quality/water-parameters

Regional programs also monitor water quality (e.g. Darwin Harbour, Great Barrier Reef water quality monitoring program).

When selecting a water quality dataset, it is essential to consider that it must include seasonal and interannual variations (i.e. at least two years), as nutrients are highly temporarily variable and tend to spike after rainfall events. The dataset must include dissolved inorganic nitrogen as NO_3^- which is the only form of N that can be directly removed from denitrification. The dataset must be as close to the site as possible and should be the primary source of nutrients into the wetland. For instance, a mangrove forest in the mouth of the river should use data sets close to the river mouth, while a mangrove forest that only is flooded by tidal water should only include datasets from marine waters.

1. The third dataset required is to determine frequency of inundation. For this step, there are three options: (a) *in situ* hydrological modelling of inundation, (b) observations from space (e.g. https://cmi.ga.gov.au/catalog/dea-water-observations-wofs, Water Observations from Space), or (c) assume tidal inundation frequency depending on vegetation type. Inundation per vegetation type can be assumed based on the tidal height predictions for the site through tidal gauges (Bureau of Meteorology: http://www.bom.gov.au/oceanography/projects/ntc/tide_tables.shtml) and local information on the height of tidal amplitude required to inundate each vegetation type. For instance, if mangroves are inundated above the 50th percentile, saltmarsh above the 75th percentile and supratidal wetlands, only during the highest tides, at the 90th percentile.

2. To estimate nutrient removal from tree uptake and carbon sequestration, measured or published literature can be obtained from national or regional studies such as Serrano et al. (2019)[34] or use values obtained from BlueCAM (Section 4.3).

3. Finally, if the goal of the restoration project was to remediate acid sulphate soils, the value of the restoration project could also be established as measurements of changes in acidity before and after the project. This can be done by analysing data from on-site monitoring programs that assess water pH and acidity (mol H+/t).

For a full discussion of water purification services measurement, please see Section 8: Water purification services.

4.5 Fish nursery services

Coastal wetlands have a well-established role as being the basis for harvestable recreational and commercial fisheries species and for coastal marine animals generally. They perform this function through two main pathways 1) as habitat in the juvenile (nursery) and adult phases, and 2) as a food source for coastal marine animals (see Figure 4.6). This section focuses on the nursery capacity of restored wetlands and their role in increasing numbers of fish and invertebrates.

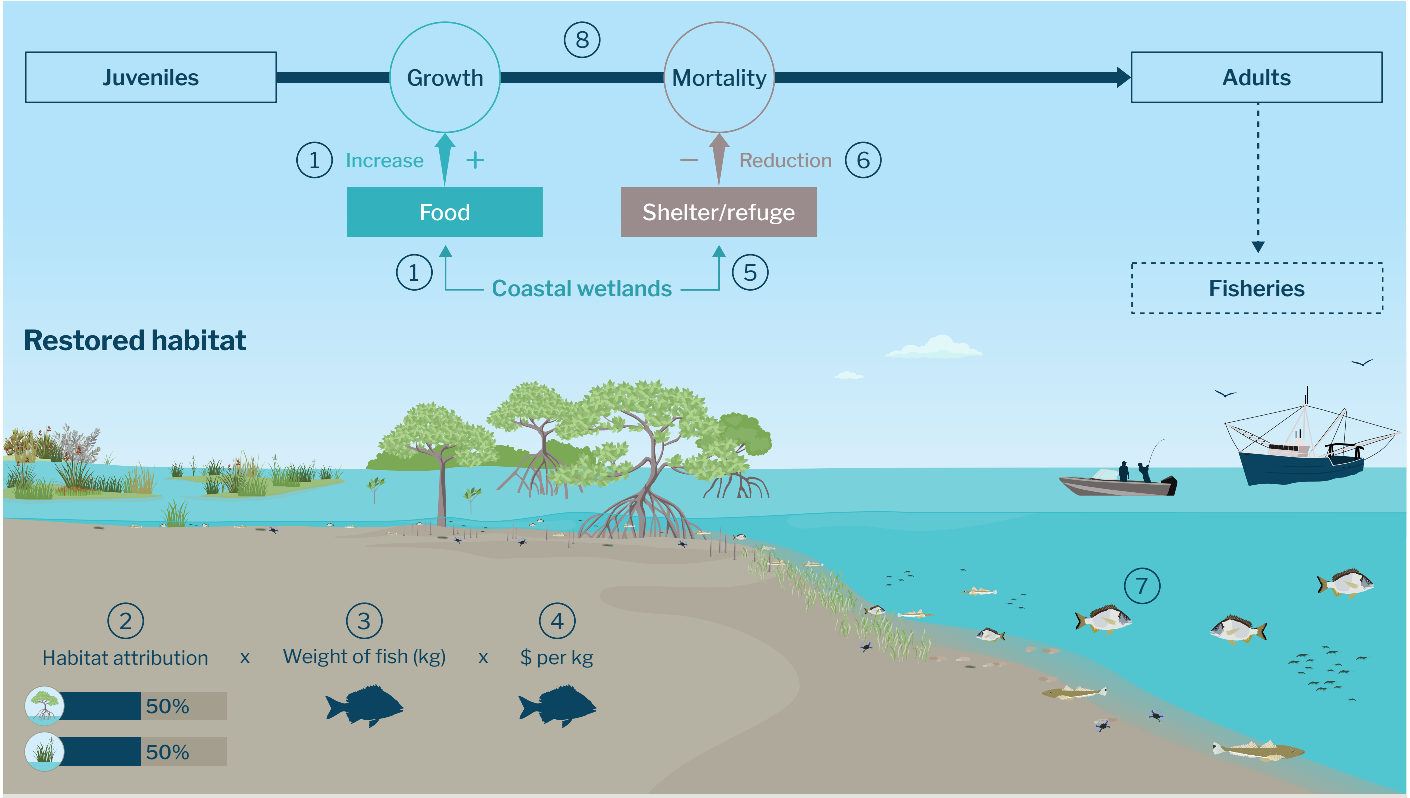
While the role of coastal wetlands in acting as nursery grounds is well established, the methods for appropriately measuring, modelling and valuing this are still being developed; there is general agreement on basic principles, but methods to value this and link it to provisioning and recreational services continue to be refined and improved. This need for further refinement stems from two issues:

1. While previous approaches have estimated the increased number of individuals or biomass of fish and invertebrates, this only represents a commercial or recreational value once the fish is caught. Therefore, this guide only reports on nursery service values as a physical service and not monetary. Additionally, the SEEA-EA guidelines use fish nursery services as an example of an intermediate service, while biomass provisioning and recreational services are the final service.

2. There are uncertainties on how much fishery production is limited by lack of available habitat rather than larval supply: at what point does increasing habitat stop increasing fish production? By collecting data from restoration projects across the country, it will be possible to begin to answer this question.

Here, the guide outlines the current best-practice method for estimating increase in fish production due to the nursery (habitat) value of blue carbon ecosystems. For fish and invertebrate species, the guide proposes measures of increased juvenile animal abundances in the different ecosystems, with subsequent modelling of contributions to the biomass of harvestable stocks (e.g. kilograms per hectare per year).

[34] Serrano O., et al. (2019). Australian vegetated coastal ecosystems as global hotspots for climate change mitigation. Nature Communications 10:4313. https://doi.org/10.1038/s41467-019-12176-8



Biomass provisioning service

1. Coastal wetlands supply food (within and beyond their borders) which is the basis of growth
2. Stable isotopes (chemical tracers) estimate the proportion each habitat contributes
3. Multiply by weight (kg) of fish caught gives the additional biomass due to a given habitat
4. Multiplied by the value ($ per kg) gives the economic contribution of the given habitat

Nursery service

5. Coastal wetlands are a source of shelter for juvenile fish
6. This decreases mortality (via predation) thereby increasing the density of juvenile fish
7. Fish leave these habitats as they grow and become available to be caught resulting in enhancement
8. Our analytic approach for this service models growth and mortality to estimate this enhancement

Figure 4.6: The nursery role of seagrass, showing the general process of using 1) field data on juvenile fish abundances, to 2) model increases in fish numbers per hectare per year and 3) convert to fish biomass per hectare per year.

Methods

Field surveys of the relative increase in abundances of juvenile animals at restored versus unrestored sites provide directly relevant data. Survey designs should use before-after and/or control-impact designs; these types of rigorous, quantitative surveys are becoming more common in restoration monitoring along with appropriate statistical analyses. Multiple unrestored control sites are encouraged even where only a single site is being restored. Modelling of enhanced contributions to adult stocks, and thus to accounts, ideally should be based on mortality rates specific to the species and to the restoration region.

For species showing differentially higher abundances in restored compared to unrestored habitat, expected contributions from juveniles to adult stocks will be modelled using known mortality rates. This provides a species-specific estimate, as well as a combined total estimate, of enhanced fisheries productivity in biomass per area per year.

Accounting for the nursery service of coastal wetlands is achieved through a four-step process:

1. Estimation of animal abundances in the different ecosystem types present in the project boundaries using direct survey methods pre-, during and post-restoration. Data from meta-analyses or collected from nearby comparable habitats may also be used where direct surveys are outside the scope of a restoration project.

2. Using the abundance data to model increases in biomass of adult stock of a) harvestable species (for commercial and recreational fishing), b) species important to cultural and First Nations ecosystem services, and c) other species of interest e.g. species of conservation significance.

3. Multiplication of the relative increase in densities due to restoration by the increase in areal extent of restored habitat; and,

4. Density enhancements (DE) for fish species from each habitat can then be estimated using the equations provided in the corresponding detailed section.

Data sources

For estimating fish nursery impacts of restoration projects, the following data will be required:

- Direct measures from field-based methods of the density of juveniles of key species (recreational, commercial, ecologically, or culturally significant) in each ecosystem type.

- Natural mortality rates from the age (size) of juveniles through to adulthood, from published literature, local fisheries departments, or publicly available data.

For a full discussion of fish nursery services measurement, please see Section 10: Fish production: Nursery habitat service.

4.6 Fish biomass provisioning services

Many years of fisheries research has provided evidence to show that restoration of coastal ecosystems can directly benefit commercial and recreational fisheries. As noted above, blue carbon ecosystems provide habitat to act as fish nurseries, but they also provide nutrition directly (for herbivores) or indirectly (via their prey that may eat marine plants) to fisheries species.

This contribution to the nutrition of commercial fisheries species can be measured by sampling the muscle tissue of different fisheries species and plants in the system and using an approach called trophic modelling to estimate the contribution of different plants (such as saltmarsh) to the diet of different species; this provides a basis for the physical accounts. This approach allows the proportion of commercially exploitable fish biomass (kg; from fisheries catch datasets) derived from a habitat to be estimated on a species-by-species basis and expressed in kilograms per hectare of habitat per year. This can then be extrapolated using habitat extend data and integrated over the time frame of the restoration project. Biomass provisioning service accounts can be turned into monetary accounts by extrapolating the value of this biomass provisioning in commercial fish markets.

Methods

Accounting can be achieved through the following steps:

1. Collect samples from fisheries and plant species to then estimate using trophic modelling, to determine how much different plant species (such as saltmarsh) contribute to the diet of commercial fisheries species. Data may also be drawn from meta-analyses if collection of relevant data is outside the scope of the restoration project.

2. Obtain extracts of annual fisheries harvest (biomass) for relevant species (kg per year) from the estuary.

3. Use the relationships in **Step 1** to partition biomass (from **Step 2**) among emergent primary producers for the system and divide by the areal extent of the relevant ecosystems (kg per hectare of habitat per year).

4. Multiply the value/s from **Step 3** by the expected or measured areal coverage of the ecosystems within the project boundaries. This provides an estimate of the commercial fisheries services that may be derived from each restored habitat type, expressed as an annual **physical account** (kg per ecosystem type per year).

5. Multiply the annual physical account by the value (AUD$ per kilogram) at the first point of sale, to derive the associated annual **monetary account** of *direct* economic output (gross value of product [GVP]) associated with the restored asset (AUD$ GVP per ecosystem type per year).

6. Scale the annual monetary account in **Step 5** using an appropriate conversion factor to estimate the total economic output [TO] associated with the restored asset (AUD$ TO per habitat type per year).

7. Deduct the costs of production (costs of fishing) from the gross value, including production inputs, labour, produced assets and intermediate inputs. This determines the annual resource rent that the fish stock provides, given the estimated biomass harvested each year.

Data sources

Data sources required to model fish provisioning services include:

1. **Stable isotope** composition of primary producers and exploited fish species within the accounting area (e.g. estuary/fishery reporting zone).
2. Commercial **fisheries catch data** at first point of sale.
3. Estimate of the total **areal extent of all ecosystems** within the accounting area (beyond project boundaries, generally estuary scale).
4. The areal extent of ecosystems in the project boundaries (see extent account).
5. Scaling biomass estimates of indirect economic activity flowing from commercial fisheries harvest in relation to the areal extent of habitats.

For a full discussion of fish biomass provisioning services measurement, please see Section 11: Fish production: Biomass provisioning service.

4.7 Coastal protection: erosion, storm mitigation and flood control services (physical & monetary)

Coastal regions in Australia are susceptible to damage from floods, erosion, and storms due to their low-lying nature and position in the coastal zone. Erosion reduction, storm mitigation, and flood control ecosystem services are grouped together as these services buffer natural processes (including natural disasters) and can reduce the damage to human infrastructure (see Figure 4.7).

Blue carbon ecosystems provide these services through a variety of mechanisms, primarily via energy absorption (e.g. wave attenuation, frictional drag, or water storage), which decreases the quantity or severity of assets at risk. The value of these services is spatially variable, due to the geographic variability of climate/bio-physical processes (the **threat**), vegetation characteristic (e.g. density or width of vegetation) (the **service**), and human infrastructure (the **benefit**).

While the three ecosystem services considered here are related, they are distinct and may not apply at every restoration site. Further, natural hazards can either be event-based, such as localised flooding or cyclonic storm surges, or associated with long-term trends, such as channel migration. While methods for quantifying these services may be similar, the processes and datasets are different. As such, the guide separately considers three services:

- Long term, persistent erosion processes (i.e. not event based).
- Flood mitigation services, associated with riverine flooding (extreme event based).
- Coastal inundation associated with storm tides, particularly coastal cyclonic events (extreme event based).

Methods

Risk is defined as the consequence of an event (e.g. houses inundated during a flood) multiplied by the likelihood (e.g. annual recurrence interval) of the event occurring. The methods used to evaluate erosion, storm, and flood mitigation services provided by blue carbon ecosystems rely on understanding (and quantifying) how wetland ecosystems can either lower the probability of an event occurring or reduce the consequence when the event occurs. While the three services included in this section need to be considered individually, Figure 4.8 shows a generic flow chart outlining an approach to assess and quantify these services.

As erosion, storm and flood mitigation services provided by blue carbon ecosystems are spatially variable, there can be significant costs associated with quantifying them. As such, the first three stages in the flow diagram will provide a low cost, rapid assessment of whether these benefits are likely to be valuable at a particular site. This assessment can be used as a screening process to assess whether further effort is justified, without significant investment or time required.

Steps 4 and 5 (Figure 4.8) relate to the physical and monetary quantification of the benefits using detailed, potentially high cost methods. The cost of this assessment will vary significantly from site to site, depending on the availability of existing data and modelling (e.g. existing flood models).

Quantification of the erosion, storm and flood mitigation benefits (Steps 4 and 5) can be completed using two pathways, namely 1) measurement or 2) process-based modelling. The appropriate application of these pathways will depend on the service being quantified.

- **Moderation of extreme events**: Process based modelling.

  Both flood mitigation and reduction in coastal inundation from storm tides (particularly cyclonic events) are services associated with extreme events. Direct measurements of these benefits can be difficult, due to the infrequent and variable nature of the processes that drive the damages. In both cases, the use of process-based models is recommended for quantifying the physical benefits. As these services are associated with a reduction in risk, a readily accepted method to quantify the monetary value is by estimating the cost of the avoided damage (using an Expected Damage Function). Avoided damages from extreme events can be calculated through changes in Annual Average Damages (AAD), measured in dollars per year.

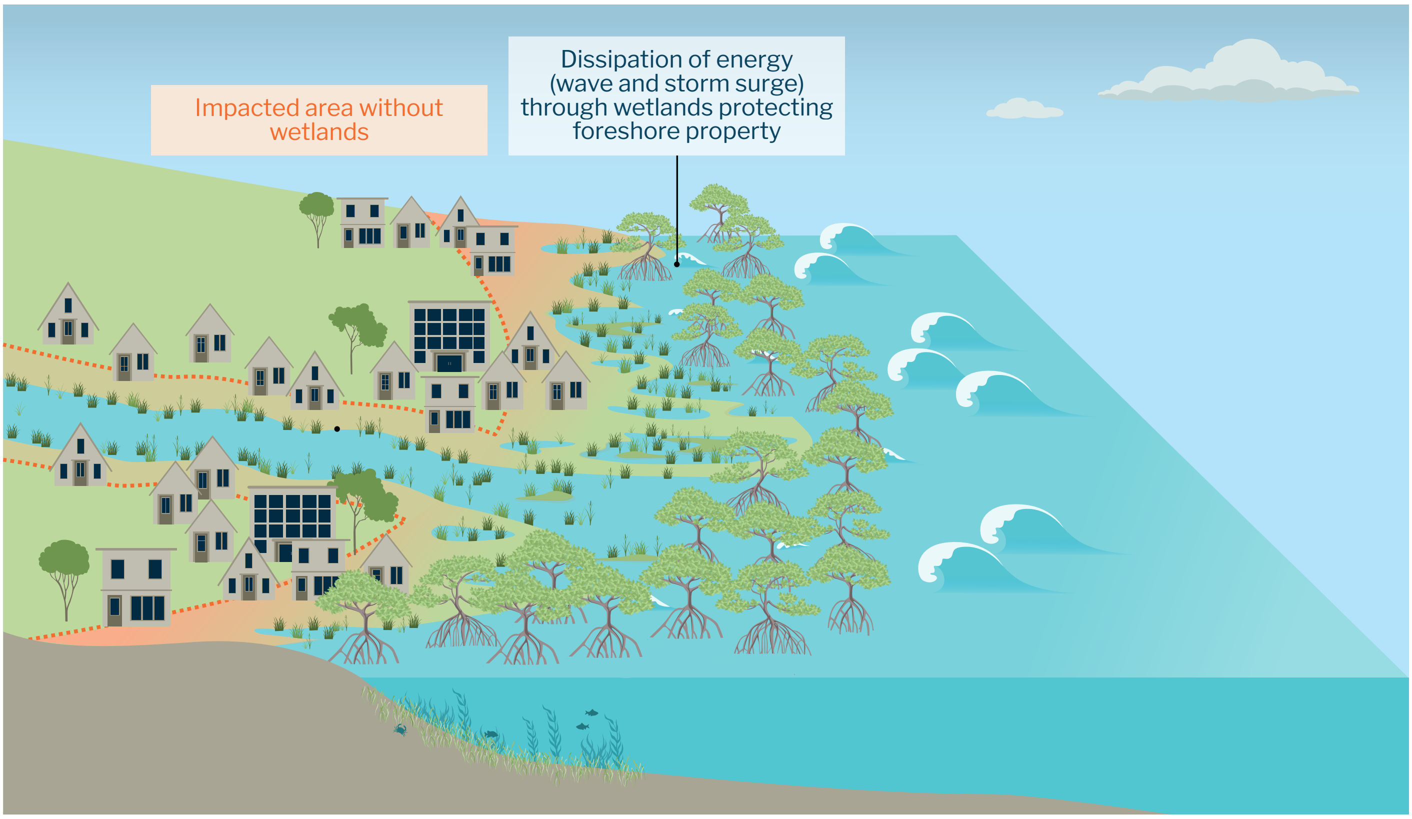


Figure 4.7: Storm surge protection service of coastal wetlands.

- **Persistent erosion**: Measurement.

  Long-term recession of shorelines, scarps or banks can be mapped and measured using field observations and/or remote sensing data. This will require sufficient data prior to restoration, which may include pre-restoration aerial imagery, to assess existing erosion trends. Following restoration, continued data collection will allow for changes (or reversal) in recession rates to be calculated. Physical accounts can be measured in area (m^2) of erosion avoided. Avoided land losses can then be valued for monetary accounts based on land valuations by the relevant state or territory valuer general.

Data sources

The data sources required (and available) will vary depending on which of the services are being considered for a project. For persistent erosion, primary data collection will be required using either field GPS measurements, or remote sensing techniques to demonstrate movement of scarps or banks both prior to and following restoration. Data on land values can typically be accessed through the relevant state or territory valuer general.

Where moderation of extreme events is to be quantified, users of the guide can assess the presence of existing process-based models, typically managed by local government authorities, for flood or coastal inundation. These models are detailed in nature and may require significant resources to build and calibrate. Use of existing models will significantly reduce the cost associated with process-based modelling, and models are likely to exist in many areas (particularly for riverine flooding) as they are used for regulatory planning. In absence of existing models, there may be substantial costs associated with quantifying protection services. As part of the methods in the detailed section, a preliminary assessment of the relevance of each environmental process may assist in understanding the potential relevance of each service prior to investment in detailed modelling. More information on data sources can be found throughout the detailed discussions on methods in the supplementary section.

For a full discussion of assessing coastal protection measurement, please see Section 9: Coastal protection: Erosion, storm mitigation, flood control services.

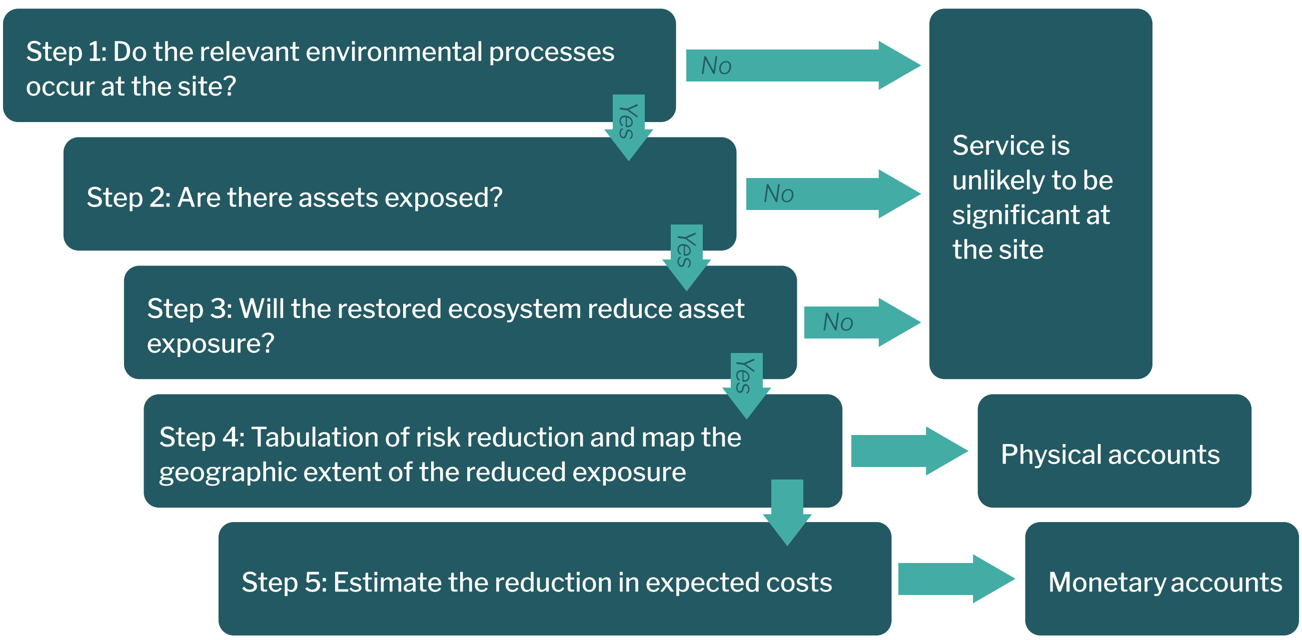


Figure 4.8: Flow chart for accounting for erosion, storm and flood mitigation services.

4.8 Cultural ecosystem services: recreation and non-use values

Cultural ecosystem services include various non-material benefits that people obtain from nature for recreational, spiritual, and psychological wellbeing. Coastal wetlands may provide cultural services such as nature-based recreation, aesthetic benefits, symbolic or spiritual benefit, as well as services that may not require use of natural assets such as the benefits derived from the knowledge that a specific natural ecosystem or wildlife exists (called 'existence values').

The health and well-being, recreational and cultural ecosystem services associated with a specific site will depend on the characteristics of the site and the human interaction with that site. Identification of the total of such services will depend on the ecological and biophysical components identified elsewhere in this Guide. This section outlines the process by which recreational and non-use cultural services (e.g. existence values) may be monetised as exchange (and, for non-use, welfare) values, as well as other economic indicators (such as job creation).

Methods

Measuring cultural service values first requires data on the service of interest and how it changes due to restoration (such as an increase in the number of fishing trips following restoration). Once this change is identified, an economic value can be estimated.

The SEEA framework focuses on 'exchange' values, typically market prices. (such as the costs incurred for a recreational fishing trip). An alternative to this is estimating 'welfare values' that reflect the benefit to consumers or producers of a good or service in addition to the cost of purchasing or producing it. However, for many cultural services associated with wetland restoration, market prices may not be possible to observe directly, as the services are freely available. The exchange values can sometimes be estimated through associated expenditures, such as the costs incurred for a recreational fishing trip. An alternative to this is estimating 'welfare values' that reflect the benefit to consumers or producers of a good or service in addition to the cost of purchasing or producing it. These can be reported in bridging tables in the accounts, and are currently the only way to represent a monetary measure of non-use values.

A range of 'non-market' valuation methods have been developed to estimate these benefits in monetary terms for situations where market prices are not available. While these methods have historically been developed to measure welfare values, they can also be used to measure exchange values aligned with the SEEA-EA in certain contexts. These include 'revealed preference' methods like the travel cost method, and 'stated preference' methods like the 'discrete choice experiment' which can estimate community values for different amounts or quality of environmental outcomes. Where primary data is lacking or budgets do not allow for the use of these methods, 'benefit transfer' can be used, which is a method of extrapolation where one draws on information from existing studies and – after making some adjustments to allow for contextual differences – transfers the available information from them to estimate a value for the site of interest.

In addition to using exchange values, EEA also compiles other measures of economic activity, recognising these are not formally measures of value. These include measures such as employment and total expenditure, and advice is given later in the Guide on how to compile this information.

The calculation of exchange and welfare values for the cultural services at a restoration site are likely to be so context-specific that it is not likely there will be existing primary data available for practitioners to use. The time, expertise and budgets required to collect new primary data are often likely to be prohibitive, meaning that a heavy reliance will be placed on the use of secondary data sources for benefit transfer.

The inevitable reliance on benefit transfer approaches means that a high degree of caution should be taken both with respect to transparently reporting and in interpreting the results from SEEA-EA of restoration projects. Primary data collection is encouraged if at all possible, and careful consideration of the confidence one can have in the results produced using secondary data is required otherwise.

Finally, in any economic valuation exercise it is important to be conscious of the risks of double-counting (elements of) the same benefits more than once. In restoration projects, this means that the concepts of additionality and substitution must be considered.

Data sources

The data required to estimate exchange and welfare values for cultural services is highly context specific. The data needs to reflect both the changes that are occurring in the environmental system of a restoration project over time, and the behaviours or preferences of the people who value that system. This combination of data needs means that it will be rare for the required primary data to already exist for a specific project site and its associated cultural services.

The collection of new primary data to evaluate cultural services will generally require implementation of a survey-based non-market valuation method. This will provide the most accurate and robust estimation of both exchange and welfare values for cultural services of restoration projects.

For smaller restoration projects, the cost of undertaking primary data collection will likely be prohibitive relative to the benefits that may arise. In these cases, benefit transfer approaches can be used, though will be less accurate. For recreation services, it is recommended to still collect primary data on visitation rates, even if the monetary valuation data is drawn from secondary sources.

Benefit transfer will require either a systematic literature search of other studies that have estimated exchange or welfare values for the same types of cultural services at similar sites, or consultation of an existing non-market value database such as The Economics of Ecosystems and Biodiversity database, Ecosystem Services Valuation Database, Environmental Valuation Reference Inventory, Environmental Valuation Database (eNvalue), the Value Tool for Natural Hazards, or the Investment Framework For Economics of Water Sensitive Cities INFFEWS Value Tool. In some cases, particularly for recreational fishing services, there are surveys conducted nationally or regionally that provide data for estimation of exchange and welfare values that can be applied to the number of fishing trips at a site, provided that the visitation rate to the project site can be calculated. In any case, we caution on using unit value transfer without adjustment, as these are prone to being incorrect when applied in a different system. If using benefit transfer, this needs to be made very clear in the forefront of any interpretation.

For a full discussion of recreational and non-use value measurement, please see Section 12: Cultural services: Recreation and non-use values.

4.9 Cultural ecosystem services: First Nations values

Establishing a way to account for Indigenous cultural values and uses within the EEA process brings unique challenges and opportunities. One challenge is that Indigenous worldviews are holistic thus their relationships with the environment are not reducible to a use or service per se, and their values are relational. EEA processes rely on technocratic approaches to socio-ecological systems that presume all components are identifiable, discrete, material and hence measurable. It is therefore problematic and possibly culturally unacceptable to separate – and quantitatively measure–- values from or traded off from each other. It is not possible to measure what is considered in Indigenous terms, the unmeasurable.

There is also no substitute for sacred goods and services. Indigenous knowledge is specific and culturally held by certain people, so how it gets treated within an EEA process needs care. It is noted that Section 2.4 in the SEAA-EA does provide recognition of the need to incorporate and recognise multiple values, and hence offers the opportunity to build on monetary valuations in alignment with SEEA-EA guidelines.

However, in Australia assessments need also to respect and reflect recognition of various cultural losses that may have occurred in the area due to colonisation. The inherent variability in Country-based value systems means a common EEA assessment process may not be appropriate, and different populations may hold different preferences/values around/for the benefits of the system. Best practice cultural accounting in Australia also assesses values not just on Indigenous owned lands but Indigenous Country.

Despite these challenges, identifying Indigenous values in an EEA process can have benefits; it can assist in Caring for Country for the relevant Indigenous group but also identify the impact and value of Indigenous cultural resource management for the system. Current EEA processes tend to focus on the flow of benefits from nature to people but does not recognise the reciprocal responsibilities of people to care for the environment, enacted by Australian Indigenous peoples via the process of Caring for Country. Cultural accounts can also help to document biocultural values in formats relevant to management.

Methods

To undertake the most effective cultural accounting process, two processes need to occur: (1) an engagement and partnership process and (2) development of the cultural account itself. The five-step approach illustrated below (Figure 4.9) is suggested as a pathway by which to implement both processes.

The development of a cultural account requires acknowledgment of cultural pluralism as well as a model that provides ways of estimating the value of ecosystems services and giving them a holistic and circular foundation rather than the atomistic and linear models that SEEA models have. There remain, however, several challenges to address along with recommendations for managing them.

Data collection is needed to map the relationship between cultural values of the site and the ecosystem benefits the Indigenous peoples derive from that site. Cultural values may be direct and indirect use values or altruistic/bequest, existence, or spiritual values. Cultural values are defined here as the importance people or groups assign to bundles of ecosystems and cultural services in a place/Indigenous Country. This includes the idea of shared values about and affiliation to Country, and whether people live within it, as this enables a meta narrative about site value that goes beyond the aggregated utilities of individuals. Co-creation of value representations is paramount. At the heart of a cultural account also, is the requirement to work out how best to render an authentic account of the interlinked values that are afforded by the case study.

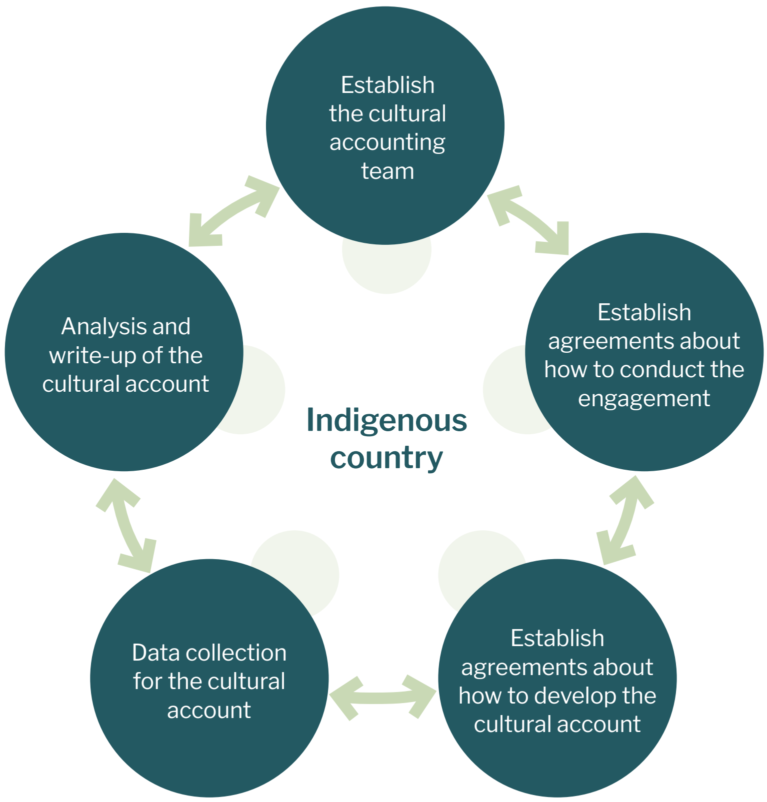


Figure 4.9: Five step pathway approach to implement engagement, partnership, and cultural account.

Data sources

Multiple data types can be collected with a range of methods, and these will depend on the nature of the restoration site and the guidance from the Indigenous partners as to what is the most culturally appropriate method to use. A combination of 2 – 3 methods is encouraged.

Desktop review:

- Archival history material.
- Church/mission records.
- Literature.
- Museum records.
- Local and other government records.
- Media and local group records.
- Anthropological and archaeological records including maps.
- Published cultural information.
- ABS data.

Participator engagement and knowledge co-production:

- On Country workshops.
- Focus groups.
- Cultural value and site mapping.
- Semi-structured interviews.
- Storytelling.
- Photo voice.

For a full discussion of Traditional Owner Cultural Services measurement, please see Section 13: Cultural services: First Nations values.

4.10 Restoration activities

Effective restoration of physical ecosystems is a pathway towards improved ecosystem service provision. On this pathway, it is important to understand the drivers of change, which are recorded in the environmental activity accounts. The physical activity accounts record and quantify activities that have taken place for the purpose of environmental protection/restoration and allow tracking of resources and efforts required to successfully restore ecosystems (see Figure 4.10).

Physical restoration activities in the context of blue carbon ecosystems can be varied, depending on the existing environment and target ecosystem (e.g. mangrove, saltmarsh, or seagrass). However, it is likely to include aspects of the following:

1. Decommissioning or modification of existing infrastructure, such as:
   - Floodgates, weirs, or other tidal barriers.
   - Levees or bund walls.
   - Artificial drainage channels.
   - Fencing.
   - Breakwaters.
2. Commissioning of new infrastructure, such as:
   - Floodgates, weirs, or other tidal barriers.
   - Levees or bund walls.
   - Artificial drainage channels.
   - Fencing.
   - Breakwaters.
3. Planting/seeding of target vegetation.
4. Management or eradication of invasive pests or weeds.
5. Chemical treatment of soil or water.
6. Labour (measured in number of days), including project and site management, and technical support.

Methods

Restoration of blue carbon ecosystems may take form via a number of different pathways, depending on the existing land use/environment and the target ecosystems (e.g. mangroves, saltmarsh, seagrass etc.). Mapping of the project area using GIS software, prior to and following restoration, is required to quantify the on-ground restoration activities.

Existing hydrological layers and high-resolution aerial imagery (see data sources) can be supplemented with on-ground site inspections utilising a handheld GPS unit (capable of accuracy of at least 5 m) to map existing infrastructure (e.g. drainage networks, floodgates, levees, or breakwaters). It is expected that all lengths and areas will be accurate within 10 m. Using the mapping created, physical activities should be tabulated, as shown in the supplementary material.

Other restoration activities that cannot be spatially mapped, such as pest animal management or labour, should be recorded directly into a table (see Table 13.9 in the detailed section). It is not expected that all aspects of Table 13.9 will to be relevant to all projects, and rows may be deleted, if required. Additional rows may be added to reflect restoration activities that have not been included in this list.

Data sources

High resolution aerial imagery and existing hydrological layers will be useful in mapping the existing site, with some relevant examples of data sources provided in the supplementary material. However, site inspections will be required in most instances to map and quantify physical restoration accounts. Site investigations should be aided by a handheld GPS unit, capable of a minimum accuracy of 5 m, to allow for spatial mapping. Other surveying methods, such as GPS enabled drones or high accuracy RTK GPS systems may also be used if the technology is available. Annual data are needed for annual accounts.

In addition to mapping, records of days worked will be required for both:

- Project and site management. This includes the time spent co-ordinating restoration works, engaging with stakeholders, consultants, and day to day management of the site. This will typically be time expended by the site owner or manager.
- Technical support and consultation. This includes the time expended to complete tasks essential for the restoration, but not explicitly associated with on-ground activities. This may include modelling, design of restoration works, or expert advice. This is likely to include time from external consultants and stakeholders.

For a full discussion of Restoration Activities measurement, please see Section 14: Restoration activities (physical accounts).

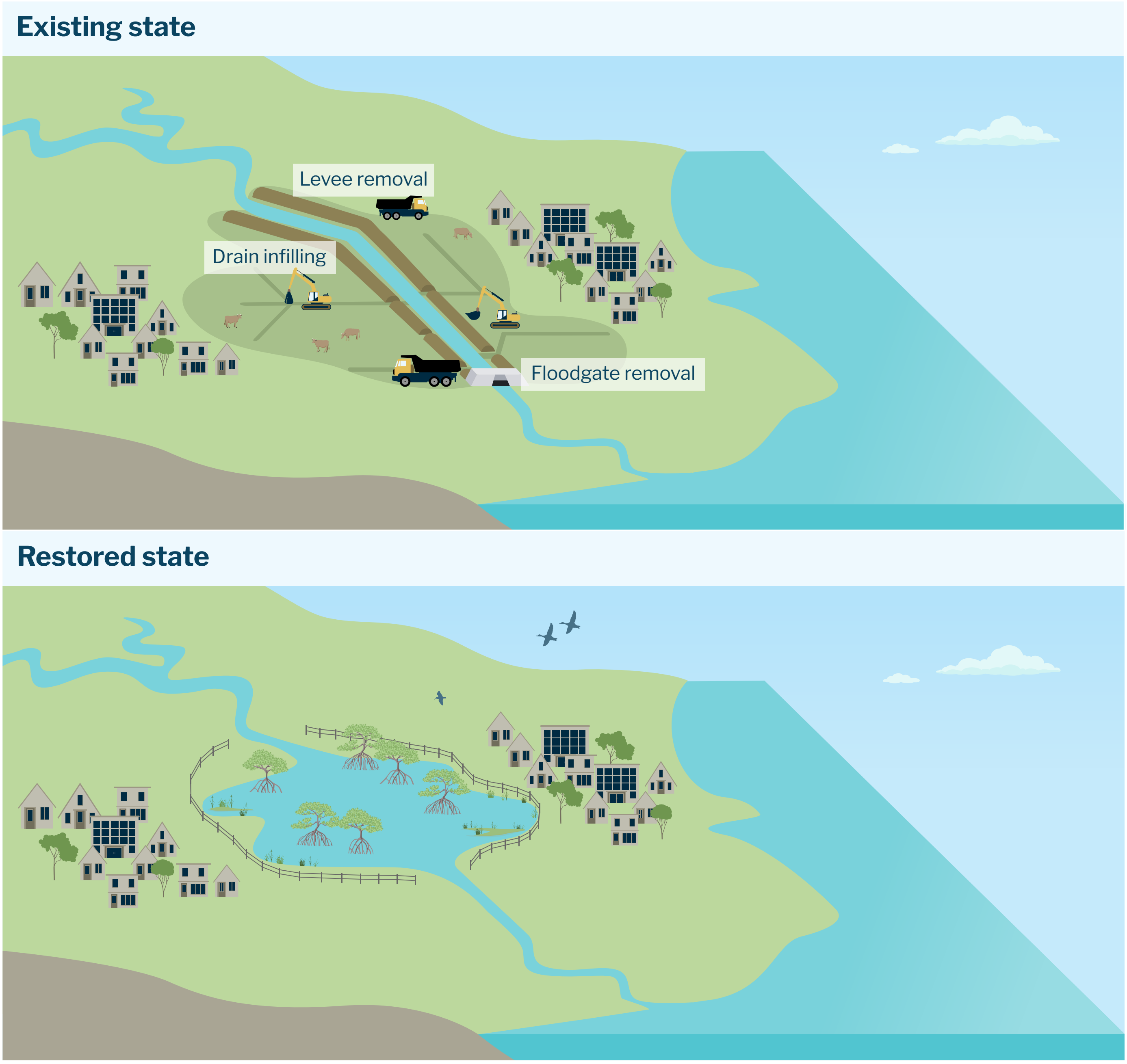
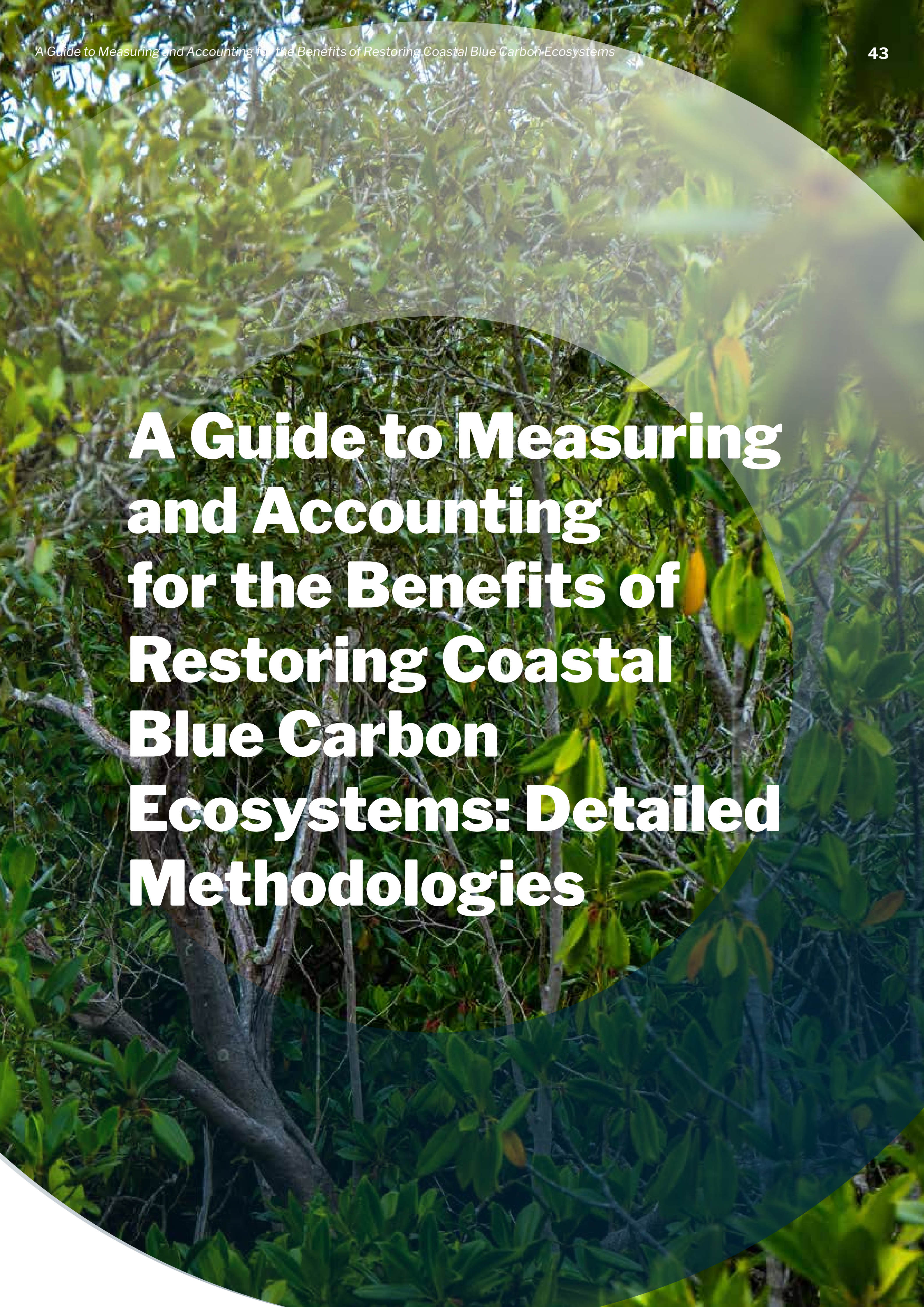


Figure 4.10: Conceptual diagram of restoration activities (such as levee removal or drain infilling).



A Guide to Measuring and Accounting for the Benefits of Restoring Coastal Blue Carbon Ecosystems: Detailed Methodologies

Authors

Paul Carnell[1], Kym Whiteoak[2], Vincent Raoult[1], Michael Vardon[3], Maria Fernanda Adame[4], Michael Burton[5], Rod M. Connolly[4], Will Glamore[6], Alice Harrison[6], Jeff Kelleway[7], Catherine E. Lovelock[8], Emily Nicholson[1,13], Melissa Nursey-Bray[9], Celeste Hill[9], Nina Wootton[9], Dewayne Mundraby[10], Dale Mundraby[10], Christopher J. Owers[11], Jacqueline B. Pocklington[1], Abbie Rogers[5], Tafesse Estifanos[5], Fitalew Taye[5], Kerrylee Rogers[7], Matthew D Taylor[12], Emma Asbridge[7], Daniel E. Hewitt[4], Peter I. Macreadie[1]

[1] School of Life and Environmental Sciences, Deakin University, VIC, Australia

[2] Canopy Economics and Policy, Melbourne, VIC, Australia

[3] Fenner School of Environment and Society, The Australian National University, Canberra, ACT, Australia

[4] Coastal and Marine Research Centre, Australian Rivers Institute, School of Environment and Science, Griffith University, Gold Coast, QLD, Australia

[5] Centre for Environmental Economics and Policy, UWA School of Agriculture and Environment, The University of Western Australia, Crawley, WA, Australia

[6] Water Research Laboratory, School of Civil and Environmental Engineering, University of New South Wales, Sydney, NSW, Australia

[7] School of Earth, Atmospheric and Life Science, GeoQuEST Research Centre, University of Wollongong, Wollongong, NSW, Australia

[8] School of Biological Sciences, The University of Queensland, St Lucia, QLD, Australia

[9] Geography, Environment, Population, University of Adelaide, Adelaide, SA, Australia

[10] Mandingalbay Yidinji Aboriginal Corporation, QLD, Australia

[11] School of Environmental and Life Sciences, College of Engineering, Science and Environment, University of Newcastle, Callaghan, NSW, Australia

[12] Port Stephens Fisheries Institute, New South Wales Department of Primary Industries, Taylors Beach, NSW, Australia

[13] School of Agriculture, Food and Ecosystem Sciences, Faculty of Science, The University of Melbourne, VIC, Australia

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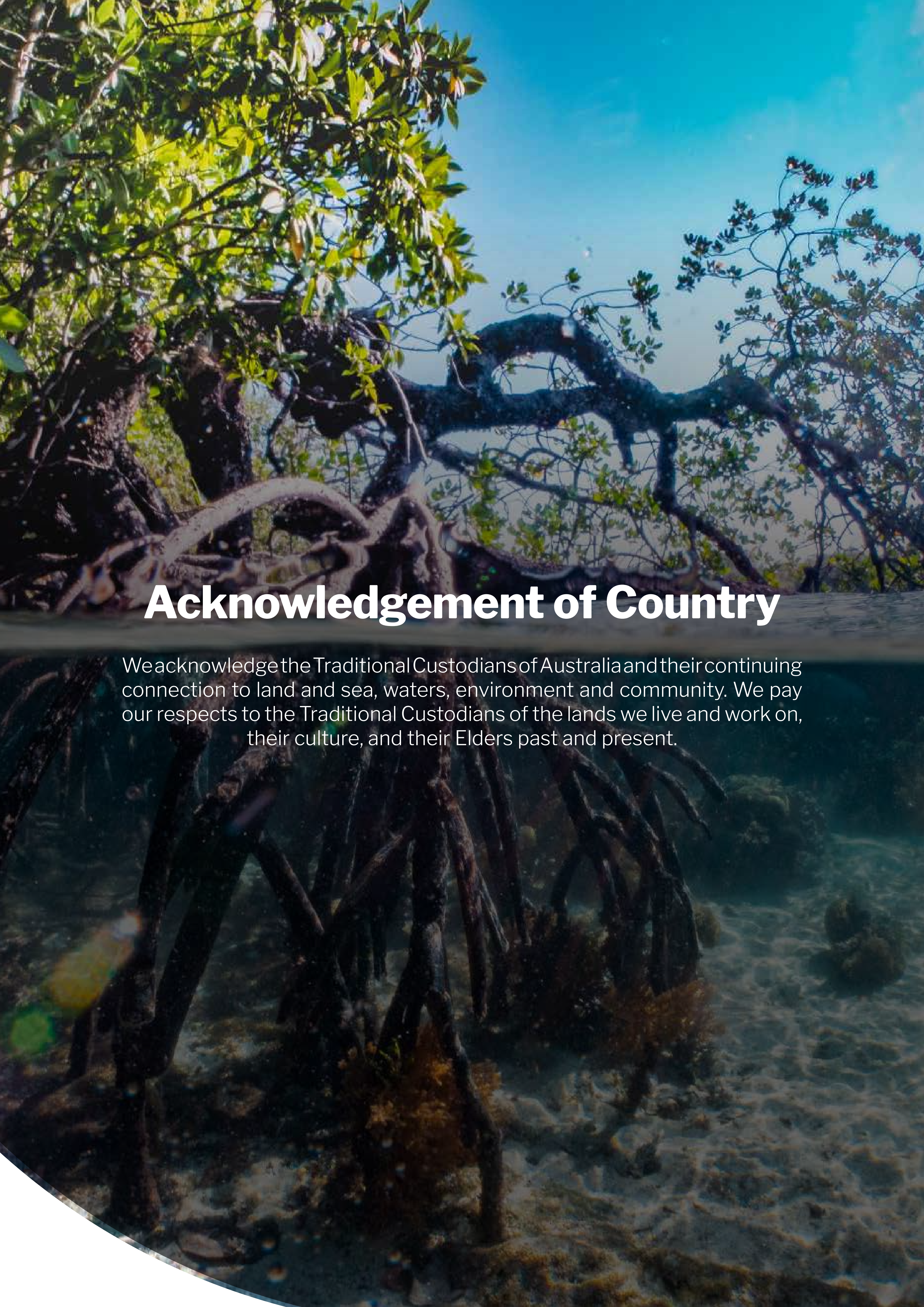

Acknowledgement of Country

We acknowledge the Traditional Custodians of Australia and their continuing connection to land and sea, waters, environment and community. We pay our respects to the Traditional Custodians of the lands we live and work on, their culture, and their Elders past and present.

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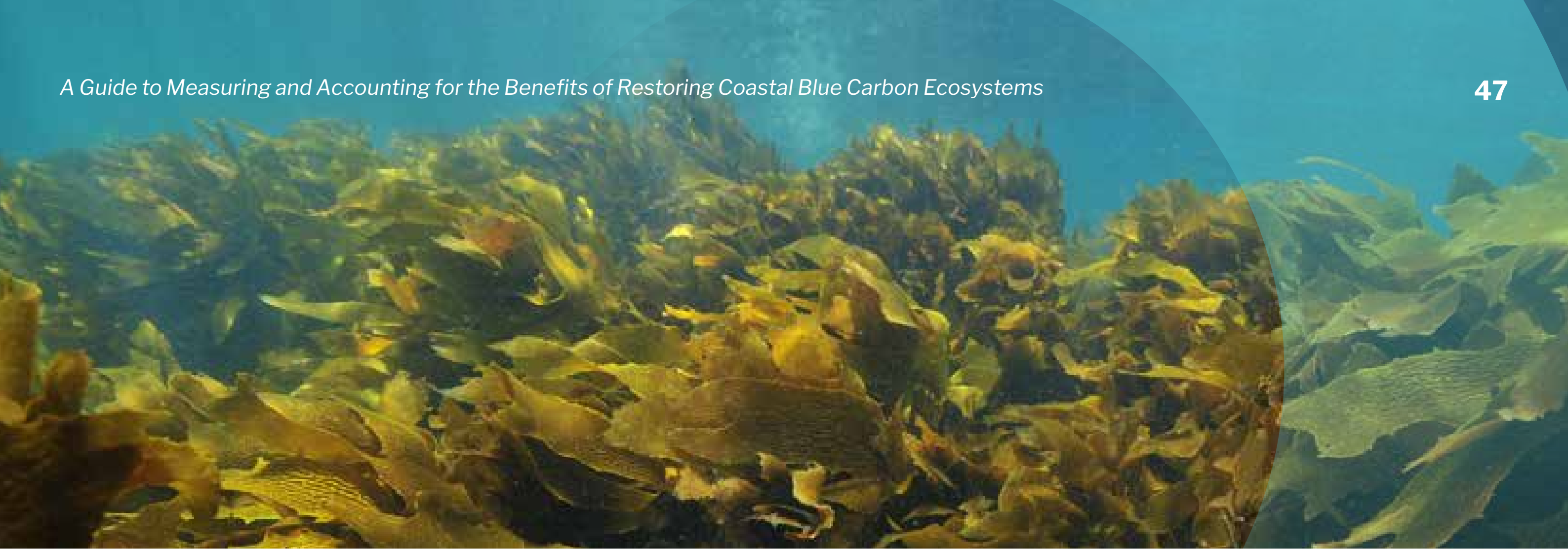

Introduction to Detailed Guide

The Guide to Measuring and Accounting for the Benefits of Restoring Coastal Blue Carbon Ecosystems presents a broad approach that uses SEEA–EA to assess the benefit of restoration projects in a rigorous, repeatable manner. The first part of the Guide provided a broad overview of the concepts of SEEA–EA and the types of methods that would be used to build accounts for extent, condition, and physical and monetary accounts. It is expected that those using this Detailed Guide have read these introductory parts of the Guide to understand the context and project scoping required before detailed methodologies are explored, or that they have previous experience designing accounts with SEEA–EA in these environments.

This Detailed Guide first explores the foundational data that underpins all the accounts that follow: the extent and condition accounts. While practitioners wishing to use this guide may only be interested in building accounts for one or a collection of ecosystem services present in this guide, the extent and condition accounts must be built prior to those ecosystem services, as various measures present in those accounts feed into the physical, cultural, and monetary accounts. The extent and condition sections are followed by a placeholder for building biodiversity accounts.

Once practitioners have built extent and condition accounts, the following sections of the guide can be used to build:

- Carbon stocks, sequestration, and emissions account
- Water purification services account
- Coastal protection and flood mitigation services account
- Fish production: nursery services account
- Fish production: biomass provisioning services account
- Cultural services: recreational and non-use values account
- Cultural services: First Nations values account
- Restoration activities: physical & monetary accounts

These sections are directed to experts in the fields of coastal wetland ecology, GIS, social science, and environmental accountants. Project proponents who have read the Guide and have committed to using the approach for their restoration project should solicit relevant expert advice to interpret the sections pertaining to the accounts they wish to build.

Foundational data – ecosystem extent & condition

Any impact assessment or set of accounts designed to quantify the impacts of a restoration project in a blue carbon ecosystem will need to develop some foundational data about the extent and condition of the ecosystems within the restoration area.

The two sections that follow, detail the proposed methodological approaches and data sources that will be needed to assemble this foundational data, pre-restoration, and then post-restoration over time.

5. Ecosystem extent

Dr Chris Owers, Dr Emma Asbridge, Dr Kerrylee Rogers

5.1 Summary of section

Ecosystem extent refers to the areal extent of different ecosystems present within an area of interest. A key measure of the success of restoration activities in blue carbon ecosystems is its change in ecosystem extent, typically measured as change from the pre-restoration extent to the post-restoration extent. An increase in areal extent of blue carbon ecosystems after restoration activities implies a good outcome where this is the goal of the project. Indicators of restoration success should target the objectives of restoration, which align with value propositions of the restoration activities (e.g. increase in saltmarsh as habitat for water birds).

Measuring coastal ecosystem extent change can be achieved using remote sensing to define vegetation community boundaries, produce maps of vegetation community distributions and quantify extents. A key component of using Earth observation data and remote sensing techniques is validation, which refers to assessing the accuracy or uncertainty of remote sensing products; this is often undertaken by comparison with analytical reference data (such as corresponding ground and field measurements or using experts to verify). Remote sensing approaches can be cost effective, reproducible, and standardised, and are effective for measuring coastal ecosystem extent and changes in extent over time. They can also provide information about biophysical and structural characteristics of coastal vegetation communities that can be useful for quantifying changes in condition (discussed further below in Ecosystem condition).

Methods

Conceptually, measuring ecosystem extent is relatively straight forward. First, ecosystem types and level of detail of differentiation need to be defined. For a simple coastal wetland restoration project this may be mangrove, saltmarsh, and grassland pastures. Secondly, these ecosystem types are delineated spatially, whereby distinct ecosystems are mapped. Finally, the delineated areas of post-restoration are subtracted from pre-restoration ecosystem extents and net change in each ecosystem extent can be quantified.

Measuring changes in ecosystem extent using Earth observation data and remote sensing technologies requires skills in spatial science. The level of experience and expertise will vary with the availability of data, and level of detail required to detect changes in ecosystem extent. Where existing national or jurisdictional products are available and appropriate for project-level ecosystem accounts, moderate skills in geographic information systems (GIS) will be required. Most national/jurisdictional products will be available in formats suitable for use in a GIS, typically raster format, and can be extracted from data portals. Digital Earth Australia (DEA) provides access to national mapping products, such as DEA mangroves, and it is anticipated that similar data products will be available for saltmarsh, supratidal forests and seagrass in the future. Using national/jurisdictional products presents a low-cost method that will provide sufficient level of detail for large restoration sites where considerable ecosystem extent change is anticipated.

Where national products are not available or not suitable for identifying project-level extent and change to extent over time, moderate to considerable GIS and remote sensing expertise will be required to generate suitable mapping products. Depending on GIS expertise available, multiple approaches can be taken to provide sufficient rigor. This approach provides capacity to undertake a more detailed approach (e.g. higher resolution data) potentially leading to greater confidence in ecosystem extent measures. A detailed approach presents a higher cost method than using existing national/jurisdictional datasets, however, may be required due to project level extents that may be at a smaller scale or where ecosystem extent change is more complex due to impact on the ecosystem or restoration intervention.

Data sources

There are a variety of Earth observation data and remote sensing technologies available for quantifying changes in ecosystem extent. Active sensors, such as Lidar and radar, provide useful information about the structure and distribution of coastal vegetation communities, whilst passive sensors provide spectral data that can be used to derive ecosystem extent. These sensors can be used from space-borne, airborne and remotely piloted aircraft; the aircraft that the sensor is affixed to and its height above the Earth will modify the resolution, precision and accuracy of remotely sensed data. Similarly, the resolution of the sensor can modify the resolution, precision and accuracy of remotely sensed data. Selection of Earth observation data and remote sensing technologies should be based on suitability and availability for mapping changes in ecosystem extent. Measuring the extent of each coastal vegetation community is important as the ecosystem services they provide vary, as do the restoration activities undertaken due to varying environmental settings and pre-impact and pre-restoration condition. Coastal vegetation communities can be differentiated based on spectral, structural and elevation range characteristics.

Mapping ecosystem extent can be achieved at multiple spatial and temporal scales. For project level environmental economic accounts, extent calculations are influenced by the resolution of Earth observation data, mapping approaches and overall accuracy of vegetation community boundaries. However, production of highly accurate maps should be balanced against the costs and expertise of production to provide sufficient rigour.

Using existing mapping products, where available, will minimise costs and expertise requirements. Approaches that can be implemented using readily available datasets that are available at a national scale will ensure standardisation of accuracy and precision. Several existing products can be used to map ecosystem extent that are publicly available. At the project-level, these datasets may not be useful to identify extent change for accounts due to the resolution of imagery, which can limit the capacity to detect changes in extent (e.g. Landsat imagery resolution of ~ 25-30 m); and the temporal availability of datasets, which can limit the capacity to measure ecosystem extent pre-impact and pre-restoration (e.g. Landsat imagery suitable for monitoring extent available from 1987 onwards).

Where existing products do not meet the needs of the project, additional data and analyses will be required. Suitable protocols and methods should be followed when using alternative data sources to ensure both suitability for the project and comparability to other project-level restoration activities.

5.2 Detailed section

Ecosystem extent refers the areal extent of different ecosystems present within an area of interest. In the case of coastal wetlands restoration, this is considering the area of blue carbon ecosystems pre-restoration as well as the area of blue carbon ecosystems post-restoration. A key measure of the success of restoration activities in blue carbon ecosystems is change in ecosystem extent. An increase in areal extent of blue carbon ecosystems after restoration activities implies a good outcome where this is the goal of the project. Indicators of restoration success will target the objectives of restoration, which align with value propositions of the restoration activities (e.g. increase in saltmarsh as habitat for water birds). In this way, assessment focuses on the change in the system (i.e. prior to restoration) and reporting based on this one location.

Ecosystem extent measures are a crucial first step for measuring the changes in blue carbon ecosystems due to restoration activities. In particular, extent measures are the foundation of environmental accounting for restoration activities and required for frameworks such as SEEA-EA. Extent measures underpin other measurements of restoration success, such as ecosystem condition, as well as providing real change in ecosystem types for identifying changes in ecosystem services. Ecosystem extent is defined in the SEEA-EA as the size of an ecosystem asset (para 2.13)[35], with the assets in this case being ecosystems within the project area. Ecosystem conversion is defined in the SEEA-EA (para 4.32)[36]; at the most basic level this specifies a conversion of 'other' ecosystems to coastal ecosystems (or vice-versa) by determining coastal ecosystem extent before and after a restoration activity. The Blue Carbon Accounting Model (BlueCAM) provides guidance for classifying each dominant ecosystem type for carbon accounting.

It should be noted that site changes may become apparent due to external drivers of change or stress not associated with the restoration activity, for example, from climate change. This would require assessments of nearby reference sites to identify if this change is region-specific or site-specific. As many factors affect restoration success, progress towards restoration goals may not occur incrementally and measurements of changes in extent over time serve as an indication of restoration success that can be monitored. Measurements of extent over time can also be useful to assess the effects of management activities or interventions. Care should be taken when monitoring changes to extent of habitat over time as there may be a delay in the response of an ecosystem to an impact or restoration activity. Measuring extent regularly (e.g. annually, every 5 years) will accommodate any temporal patterns in ecosystem response (e.g. seasonality).

Measuring coastal ecosystem extent change can be achieved using remote sensing to define vegetation community boundaries, produce maps of vegetation community distributions and measure extents. Remote sensing approaches can be cost effective, reproducible, and standardised, and are effective for measuring coastal ecosystem extent and changes in extent over time[37,38]. They can also provide information about biophysical and structural characteristics of coastal vegetation communities that can be useful for quantifying changes in condition (discussed further Section 6: Ecosystem condition).

There are a variety of Earth observation data and remote sensing technologies available to measure changes in ecosystem extent. Active sensors, such as Lidar and radar, provide useful information about the structure and distribution of coastal vegetation communities, whilst passive

[35] United Nations, et al. (2017). System of Environmental-Economic Accounting 2012: Central Framework. Manuals & Guides. United Nations. https://doi.org/10.5089/9789211615630.069

[36] United Nations, et al. (2017). System of Environmental-Economic Accounting 2012: Central Framework. Manuals & Guides. United Nations. https://doi.org/10.5089/9789211615630.069

[37] Owers, C. J. et al. (2022). Operational continental-scale land cover mapping of Australia using the Open Data Cube. International Journal of Digital Earth, 15(1), 1715-1737. https://doi.org/10.1080/17538947.2022.2130461

[38] Nagendra, H. et al. (2013). Remote sensing for conservation monitoring: Assessing protected areas, habitat extent, habitat condition, species diversity, and threats. Ecological Indicators, 33, 45-59. https://doi.org/10.1016/j.ecolind.2012.09.014

sensors provide spectral data that can be used to derive ecosystem extent. These sensors can be used from space-borne, airborne, and remotely piloted aircraft, and this will affect the resolution, precision, and accuracy of remotely sensed data. Selection of Earth observation data and remote sensing technologies should be based on suitability and availability for mapping changes in ecosystem extent. Measuring the extent of each coastal vegetation community is important as the ecosystem services they provide vary, as do the restoration activities undertaken due to varying environmental settings and pre-impact and pre-restoration condition. Coastal vegetation communities can be differentiated based on spectral, structural and elevation range characteristics.

A key component of using Earth observation data and remote sensing techniques is validation, which refers to assessing the accuracy or uncertainty of higher-level remote sensing products with analytical reference data (such as corresponding ground and field measurements or using experts to verify mapping). Validating ecosystem extent measurements does increase cost, however, is directly proportional to level of confidence in extent estimates. In particular, rapidly changing land uses associated with restoration efforts can be easily misclassified using remote sensing techniques, and ground and field measures of validation will be preferable to quantify reliability of extent estimates.

There is potential to map and quantify the extent of transition areas between two adjacent ecological communities (ecotones), where appropriate, in detailed site level assessments; however, this depends on the availability of field based and very high-resolution Earth observation data. If this data is not available, a national scale approach should be used to map ecosystem extent which will limit the ability to accurately define ecotones. There is a need to map the extent of land cover/ land use categories, such as urban and agriculture, to track change over time and how this affects blue carbon ecosystem extents.

A workflow of decisions prior to delineating the extent of blue carbon landcover units at pre- and post-restoration periods is detailed below (Figure 5.1). Assessments of extent should always be defined in the context of the goals of restoration and should occur within the confines of the focal area of interest, typically defined by the extent of the restoration project. The vegetation type to be delineated can then be defined based on the goals for restoration and the land cover units within the focal area of interest. For example, if the goal was to increase saltmarsh area, then delineating saltmarsh extent pre- and post-restoration within the restoration area are critical; if the goal was to increase saltmarsh extent and have no net loss in blue carbon ecosystems, then the ‘other’ blue carbon ecosystems should also be delineated. The approach for delineating blue carbon land cover units, and other land cover units then needs to be determined.

There is benefit in using national products that are publicly available and do not require generation of new land cover products (a decision flow chart is given in Figure 5.2). For example, lower levels of GIS and remote sensing skills and knowledge are required. In addition, extent can be defined relatively efficiently as new products do not need to be generated. Where national products are not available, an alternative approach will need to be established for delineating blue carbon land cover units. In some instances, national products may be available but will not be sufficient for assessing extent of blue carbon units within the area of interest or with respect to the goal of restoration. This is determined initially by ascertaining whether the national products can be validated for the pre- and post-restoration periods. This can be undertaken on the basis of site-specific knowledge, comparison with other datasets, or preferentially, by validating the extent of blue carbon land cover units pre- and post-restoration. If the datasets do not validate well, then it is appropriate to choose an alternative approach for delineating blue carbon land cover units. If the national products can be validated, the products then need to be assessed as to whether they are suitable for assessing extent within the area of interest. For example, it may not be appropriate to use a national product with medium resolution (e.g. Landsat derived products) to quantify extent within a small restoration area, but it is appropriate to use a national product of medium resolution to quantify extent of blue carbon units within an area of interest that is expansive. If the resolution of the national product is suitable for assessing extent within the area of interest, then the national product needs to be assessed to determine whether it sufficiently delineates the target ecosystems for restoration, and if they do not, the alternative approaches need to be identified for delineating the blue carbon land cover units that are the target of restoration. Providing national products are available, the products are available at a scale corresponding to the scale of the restoration product, and the national products delineate the blue carbon ecosystems that are the goal of restoration, then national products can be used to quantify extent. Otherwise, an alternative approach for delineating blue carbon land cover units needs to be established. There are a range of approaches that can be used and many different remotely sensed products that can be used as input data, and the decision regarding the remotely sensed data and approach should be determined on the basis of a benefit-cost analysis. Moreover, the selected approach should allow blue carbon land cover units to be delineated sufficiently and balance against the costs of delineating extent.

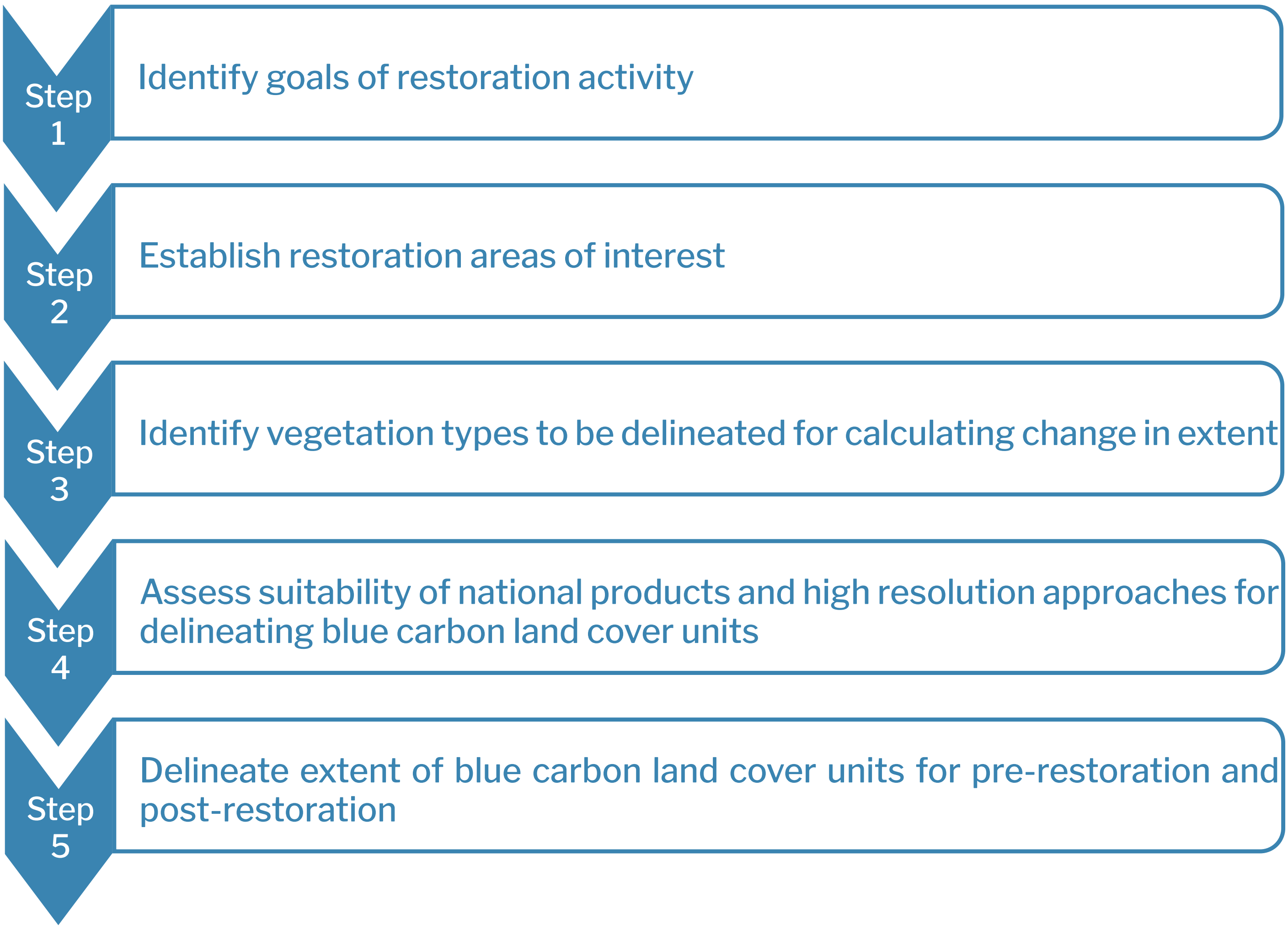

Figure 5.1: Steps undertaken prior to delineating extent of blue carbon land cover units at pre- and post-restoration time steps.

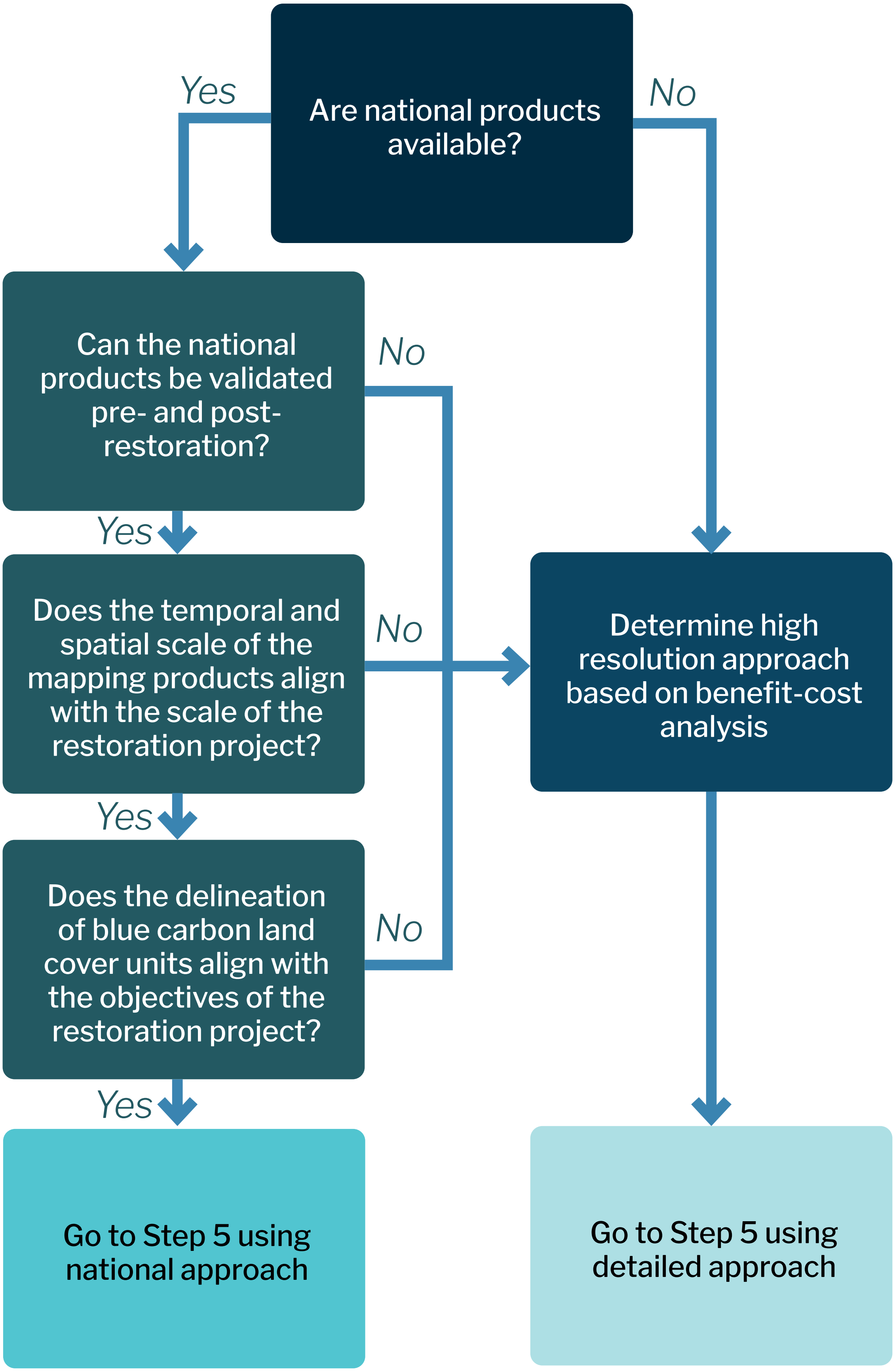

Figure 5.2: Decision flow-chart for determining whether national products can be sufficiently used to assess the change in extent of blue carbon land cover units.

Data sources

Mapping ecosystem extent can be achieved at multiple spatial and temporal scales. For project level environmental economic accounts, extent calculations are influenced by the resolution of Earth observation data, mapping approaches and overall accuracy of vegetation community boundaries. However, production of highly accurate maps should be balanced against the costs and expertise requirements of production to provide sufficient rigour.

Using existing mapping products, where available, will minimise costs and expertise requirements. Approaches that can be implemented using readily available datasets that are available at a national scale will ensure standardisation of accuracy and precision. Several existing products can be used to map ecosystem extent that are freely available. At the project-level, these datasets may not be useful to identify extent change for accounts due to the resolution of imagery, which can limit the capacity to detect changes in extent (e.g. Landsat imagery resolution of ~ 25-30 m); and the temporal availability of datasets, which can limit the capacity to measure ecosystem extent for natural state and pre-restoration (e.g. Landsat imagery suitable for monitoring extent available from 1987 onwards).

Where existing products do not meet the needs of the project, additional data and analyses will be required. Suitable protocols and methods should be followed when using alternative data sources to ensure both suitability for the project and comparability to other project-level restoration activities.

Exisiting mapping products

Using existing mapping products, where available, will minimise costs and expertise requirements. Approaches that can be implemented using readily available datasets that are available at a national scale will ensure standardisation of accuracy and precision. Several existing products can be used to map ecosystem extent that are freely available. These include (but are not limited to):

- **Mangroves**: Digital Earth Australia (DEA) Mangroves, providing annual mangrove extent for Australia at 25 m resolution from 1987-2018[39].
- **Saltmarsh**: An Australia-wide product is currently in development, providing similar spatial and resolution as DEA Mangroves, led by researchers at JCUs Global Ecology Lab[40]. State-wide data is available for some states (e.g. NSW Fisheries Data Portal), and these are collectively available through Seamap Australia[41].
- **Supratidal forests**: An Australia-wide product is currently in development, providing similar spatial resolution as DEA Mangroves. Global Forestwatch data may also be useful for these purposes.
- **Seagrass**: No national map currently exists for seagrass mapping, largely due to the distinct challenges of mapping ecosystem extent under water (see limitations section below). The National Environmental Science Program is scoping feasibility of undertaking national scale seagrass mapping. State-wide data is available for some states (e.g. NSW Fisheries Data Portal, CoastKit Victoria[42]), and these are collectively available through Seamap Australia[43] .
- **Tidal flats**: Global intertidal change provides a global tidal flats change product[44].
- **'Other' ecosystems**: DEA Land Cover, providing annual land cover information (> 100 land cover attributions) for Australia at 25 m resolution from 1988-2020[45].

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[39] Accessed at: https://cmi.ga.gov.au/data-products/dea/191/dea-mangrove-canopy-cover-landsat

[40] Accessed at: https://www.saltmarshes.org/

[41] Accessed at: https://seamapaustralia.org/

[42] Accessed at: https://mapshare.vic.gov.au/coastkit/

[43] Accessed at: https://seamapaustralia.org/

[44] Accessed at: https://www.intertidal.app/

[45] Accessed at: https://cmi.ga.gov.au/data-products/dea/607/dea-land-cover-landsat

These data sources, where developed, provide readily accessible extent of coastal ecosystems, some with temporally resolutions spanning the past 30 years and will facilitate a national approach to environmental economic accounts. Depending on the project-level restoration area, these datasets may not be useful to identify extent change for accounts due to the resolution of imagery, which can limit the capacity to detect changes in ecosystem extent (e.g. Landsat imagery resolution of 30 m); and the temporal availability of datasets, which can limit the capacity to measure ecosystem extent of natural state and pre-restoration (e.g. Landsat imagery suitable for monitoring extent available from 1987 onwards). Other data sources may be required to provide sufficient rigour that aligns with project-level knowledge and specifications.

Mapping collation and/or data processing

Where existing products do not meet the needs of the project, additional data and analyses will be required. This can be undertaken using the following data sources:

- Existing mapping products of ecosystem extents[46].
- Historical aerial photography.
- Recent high-resolution satellite or airborne imagery (e.g. Planet or Nearmap).
- Land use information (e.g. local council cadastral information, ABARES products).
- Airborne Lidar data.
- Imagery and Lidar data from Remote Piloted Aircraft (RPAs) to produce orthomosaics, Digital Elevation Models (DEMs), Digital Surface Models (DSMs) and Canopy Height Models (CHMs).
- Spaceborne C- and L-band radar (e.g. Sentinel 1, ALOS PALSAR) and optical sensors (e.g. Landsat).

Suitable protocols and methods should be followed when using alternative data sources to ensure both suitability for the project and comparability to other project-level restoration activities. Use of alternative data sources will require greater expertise to produce appropriate maps of ecosystem extent.

Methods

Where national products are available and appropriate for project-level ecosystem accounts, skills in geographic information systems (GIS) will be required. Most national products will be available in formats suitable for use in a GIS, typically raster format, and can be extracted from data portals. Digital Earth Australia (DEA) provides access to national mapping products, such as DEA mangroves, and it is anticipated that similar data products will be available for saltmarsh, supratidal forests and seagrass in the future. For example, DEA mangroves can be used to quantify mangrove extent at the project-level, providing the following can be resolved:

1. Area of interest (latitude and longitude) and time period (pre-restoration, post-restoration years) established.
2. Data for area of interest extracted out using DEA data portal.
3. Data loaded into GIS software (e.g. QGIS, ArcGIS).
4. Mangrove areal extent calculated using zonal statistics tools.

Where national products are not available or not suitable for identifying project-level extent and change to extent over time, moderate to considerable GIS and remote sensing expertise will be required to generate suitable mapping products. Depending on GIS expertise available, multiple approaches can be taken to provide sufficient rigor. For example, steps to extract coastal wetland extent from analysis-ready digital airborne imagery includes:

1. Identifying and extracting the restoration area of interest using a mask (i.e. polygon or rasterised object indicating area of interest).

[46] See for example http://www.ozcoasts.gov.au/geom_geol/vic/index.jsp

2. Extract data on mangrove area, for example, using a mask of delineated ecosystem extent (i.e. polygon or rasterised object indicating ecosystem extent).

3. Calculate and sum the area of all polygons.

4. These approaches are dependent on data suitability and GIS expertise. Approaches used will also provide different degrees of accuracy and precision. Providing expertise is appropriate and consistent approaches are used, change in coastal ecosystem vegetation community extent can be quantified at an appropriate level for project-level environmental economic accounts. A worked example is provided in two case studies[47,48].

Key assumptions or limitations

Ecosystem extent accounts will provide fundamental information for assessing success of restoration activities and support development of environmental economic accounts. For this reason, accuracy and precision of estimates should be prioritised. A key assumption is the familiarity of the project team to coastal ecosystem characteristics as evident in imagery (i.e. what do mangrove and saltmarsh look like in different settings, where are they likely to be present). If expertise in delineating coastal ecosystem vegetation communities is limited, the accuracy of extent calculations will be affected, and extent calculations may be erroneous.

These errors can increase when vegetation communities have broad ecotones, or communities are adjusting to changes in environmental conditions. In many cases, vegetation communities in coastal ecosystems can only be separated into units (i.e. mangrove, saltmarsh, seagrass) with low precision as they are distributed along a continuum of environmental conditions (i.e. proportion of cover of mangrove or saltmarsh in a pixel); this reduces the capacity to identify nuances in successful restoration activities (e.g. increase in saltmarsh cover under a supratidal swamp forest). Irrespective of these difficulties, delineating the extent of coastal ecosystem vegetation communities is important for monitoring, accounting, and reporting (e.g. varying capacity for carbon storage or biodiversity benefits).

Data availability and suitability may influence the selection of methods for quantifying vegetation community extent pre- and post-restoration. Factors that may influence capacity to delineate boundaries and quantify extent include:

- **Data not fit for purpose**: For example, cloud cover is common in imagery for coastal ecosystems throughout the tropics. This is particularly problematic when using passive satellite sensors such as Quickbird, WorldView, Landsat and MODIS. However, this can be somewhat resolved by using analysis ready data (e.g. DEA products), using active sensors, selecting cloud free scenes or by applying a cloud mask during processing. In addition, high-resolution data may not be available prior to restoration, thereby preventing pre- and post-restoration comparison. This could be overcome by using historic aerial photography or using the dense time-series of Landsat data available extending back to 1987. High resolution data (i.e. Planet, WorldView etc.) is expensive, and this limits its application in many projects. For detailed local-scale assessments remotely piloted aircrafts (RPAs i.e. drones) can be used to provide high-resolution aerial imagery; this can allow for periodic and consistent data capture as part of a long-term monitoring program. In addition, field based on-ground surveys may be useful as part of the detailed local assessments to verify remote sensing analyses and capture high resolution information. Field data collection can only be carried out after considering feasibility and the need for highly detailed information. For seagrass extent mapping, the clarity of the water column influences

[47] Glamore, W., et al. (2023). Accounting for benefits from coastal restoration: a case study from the Hunter River. Report to DCCEEW.

[48] Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW.

the capacity to distinguish seagrass using remote sensing. Other factors influencing clarity and spectral reflectance signals include inundation with tides (short stature mangroves and saltmarsh may be submerged), waves, water turbidity and colour, sun glint, shadows and presence of epiphytes covering vegetation.

- **Cost**: While some satellite and remotely-sensed products are free such as through the DEA datacube, Sentinel 2 and Landsat, other higher resolution aerial imagery such as NearMaps or WorldView often comes at a cost. While you will need to obtain a quote specific to your project, we have provided indicative prices in Table 5.1 below.

- **Variable spectral signatures**: For example, vegetation communities may occur in a range of densities and the variable influence of the underlying substrate and tides on spectral signatures is difficult to characterise. To address these limitations, low tide composites could be used to remove the uncertainties associated with inundation and wet or dry underlying substrate. This can be achieved using the DEA high and low tide imagery product, of cloud free composite imagery at high and low tides.

- **Limited GIS expertise** will also influence the capacity to produce reliable project-level mapping. This may be addressed by using readily accessible data products and guided tutorials such as DEA notebooks (https://github.com/GeoscienceAustralia/dea-notebooks). This will influence the reliability and accuracy of extent calculations.

- **Other habitats**: Lack of existing national map products for other habitat types such as seagrass, kelp, and oyster reefs (extent mapping is largely incomplete with varying methods) limits the capacity for mapping extent. Detailed on-ground assessments may be necessary in some cases, however, this may also be limited by clear observations (e.g. water clarity, high tide). A solution may be field-based assessments done pre- and post-restoration activities, when the water quality is clear and turbidity is at a minimum (i.e. following periods of dry weather with reduced wind and wave speeds). The approach used by McKenzie et al.[49] to map seagrass offers a useful insight as field-based assessments were prioritised for validation of the UNEP World Conservation Monitoring Centre (WCMC) database and additional seagrass mapped polygons in the public domain. However, these field-based assessments would need to be considered in terms of their feasibility and the need for on ground verification.

[49] McKenzie, L. J. et al. (2020). The global distribution of seagrass meadows. Environmental Research Letters, 15(7), 074041. https://doi/org/10.1088/1748-9326/ab7d06

Uncertainties

Depending on the approach used to map extent of ecosystems, there are different ways in which uncertainties enter into extent accounts.

1. When using existing datasets that may be ecosystem specific (eg. The national mangrove map and a state based seagrass map), it may be that an individual pixel is mapped as two different (or more) ecosystem types. This may be due to the resolution of the different mapping products (eg. One may be finer resolution (eg. Using Sentinel 2 at 10 m resolution), while the other may be coarser resolution (eg. Using Landsat at 30 m resolution). In this instance the project team will have to make a decision based on their own knowledge of the datasets and the project site as the best way to deal with this issue.

2. be less certain if it is one ecosystem type or another. In this instance, it is possible to run the ecosystem predictor model multiple times (for example 10), and then report for each pixel the number of times the same ecosystem type is predicted for that pixel (for example, a area with higher certainty would be predicted as the same ecosystem type 10 out of 10 times, which an area with lower certainty would be predicted as one ecosystem type 5 out fo 10 times (effectively a 50/50 split). In this way, we can provide a measure of certainty to sit behind the extent maps that are produced.

Table 5.1: Detailed information on type, source, years available, resolution, bands, and price (if relevant) for satellite and aerial imagery from different sources. Prices are indicative only, and an individual quote must be sought for your project area.

Image Source	Years available	Resolution	Publicly available (yes/no)	Indicative Price
WorldView-2	since 2009	Panchromatic imagery of 0.46 m; multispectral imagery with 1.84 m	No	US$17.50 per km^2, usually minimal order size of 25 km^2
WorldView-3	since 2014	Panchromatic imagery at 0.31 m and multispectral imagery at 1.24 m	No	US$22.50 per km^2, usually minimal order size of 25 km^2
Landsat-8	since 2013	Panchromatic imagery of 15 m; multispectral imagery with 30 m	Yes, http://www.usgs.gov/	Free
Landsat-4	1982 - 1993	MSS: 60 m; TM: 30 m, but thermal infrared at 120 m resampled to 30 m pixels	Yes, http://www.usgs.gov/	Free
Landsat-5	1984 - 2013	MSS: 60 m; TM: 30 m, but thermal infrared at 120 m resampled to 30 m pixels	Yes, http://www.usgs.gov/	Free
IKONOS-2	1999-2014	1 m panchromatic, 4 m multispectral	No	US$10 per km^2, usually minimal order size of 25 km^2
QuickBird	2001 - 2014	Panchromatic imagery at 65 cm and multispectral at 2.62 m	No	US$17.50 per km^2, usually minimal order size of 25 km^2

Table 5.1: cont.

Image Source	Years available	Resolution	Publicly available (yes/no)	Indicative Price
Sentinel 2	since 2015	10, 20, 60 m	Yes http://docs.dea.ga.gov.au/setup/sandbox.html	Free
ZY-3A	since 2012	5.8 m	No	Not publicly available
RapidEye (Planet Lab)	2008 - 2020	5 m	No	Not publicly available
SPOT 1, 2, 3	1986 - 2002	20 m MSS, 10 m Pan	No	?
SPOT 4	1998 - 2013	20 m MSS, 10 m Pan	No	?
SPOT 5	2002 - 2015	10/20 m MSS, 2.5 m Pan	No	?
SPOT 6, 7	since 2012	6 m MSS, 1.5 m Pan	No	US$4.75 per km^2, usually minimal order size of 100 km^2
Nearmap	since 2008	5.8 - 7.5 m	No	Different plans available. A small commercial licence is around $500

6. Ecosystem condition

Dr Paul Carnell, Dr Jacqueline B. Pocklington, Prof. Kerrylee Rogers, Dr Emma Asbridge, Prof. Emily Nicholson

6.1 Summary of section

Ecosystem condition is key to restoration planning, implementation, and for monitoring and evaluation of restoration success. The condition of an ecosystem is dependent on the abiotic conditions which can be split into physical (e.g. temperature, , hydrology) and chemical (e.g. salinity, pH) and how they may have changed to impact the plants and animals that inhabit the ecosystem. The condition of the ecosystem is not just the condition of the habitat forming plants or animals (e.g. mangroves), but also the diversity and abundance of the species in the system (compositional), the structural components, ecosystem function (e.g. productivity and predation) and connectivity (seascape characteristics)[50].

Ecosystem condition is defined in the SEEA-EA as 'the quality of the ecosystem measured in terms of its abiotic, biotic and landscape/seascape characteristics (para 2.13[51]). The approach for measuring the condition of ecosystems follows a similar approach developed for the IUCN Red List of Ecosystems and applied in SEEA-EA as the Ecosystem Condition Typology[52]. Measures of condition are ecosystem-specific and should have a conceptually similar reference baseline as the basis for developing indicators of condition. SEEA-EA guidelines recommend that the reference state should be 'natural'; in an Australian context, this is typically an estimate of pre-European colonisation state, often based on example sites representing 'best of what's left', or estimates or models of what this could have been (but see step 5 below for more detail). Setting this reference state should be undertaken carefully and may require a stakeholder/or expert elicitation process within the context of the goals of the project. A 'natural state' should also refer to a reference condition based on the principle of maintaining ecosystem integrity, stability, and resilience over ecological timeframes.

While the 'natural state' sets the upper level for condition assessments, this may be quite different to the restoration goal for a given site. Recognising that it may not be possible to restore an ecosystem back to its 'natural state' due to the influence of dynamic processes such as sea-level rise, changes in rainfall, or declines and extinction of species. Therefore, we recommend making it clear in the condition account where the restoration goal may be different to the reference level.

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[50] Carey, J., et al. (2017). Report on Condition of Yaringa Marine National Park - 2002 to 2013. Parks Victoria Technical Series No. 112. Parks Victoria, Melbourne.

[51] United Nations et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

[52] Keith, D.A., et al. (2020). The IUCN Global Ecosystem Typology 2.0: https://portals.iucn.org/library/sites/library/files/documents/2020-037-En.pdf

Quantifying ecosystem condition can be challenging, due to the complexity of ecosystems, and the diversity of purposes for condition accounts. Importantly, scope to capture changes that affect the intrinsic values of the system should be included, such as biodiversity and cultural values. Condition variables should aim to capture a broad range of ecosystem attributes, such as biotic and abiotic aspects, structure, function and composition, and land/seascape or contextual factors (e.g. fragmentation). Because different ecosystems can have different structures, such as mangrove trees and saltmarsh grasses, the measures of ecosystem condition are ecosystem (can be many within same site) and project specific. This means that indicators of condition may vary between ecosystems depending on what 'good' condition may be for that ecosystem and for a restoration project. Nevertheless, ecosystem condition can be quantified using remote sensing approaches, modelling, and field measures. Practitioners and experts in condition assessment should be engaged to do this. Categorical or continuous maps can be developed indicating condition scores and spatial variation, and through analysis and modelling of field-collected data.

6.2 Detailed section

Selection of variables as indicators of ecosystem condition

Condition accounts are made up of ecosystem-specific condition variables that cover many ecosystem attributes (composition, structure, and function, as well as landscape context and connectivity, across biotic and abiotic components of the ecosystem), from diverse data sources, including field-based data, remotely-sensed data, expert judgement and modelling. Including data from many sources increases complementarity of information, covers multiple dimensions of ecosystem condition, and will include different values and capacity to sustain various services. Expertise in data integration and validation is required to do this in a way that ensures robust and defensible outcomes.

Describing the ecosystem, its key features, attributes, and processes is critical for characterising the ecosystem prior to building accounts. This is why developing a conceptual model of the ecosystem that depicts relationships between the identified features, and key processes and threats is a key step in building accounts, especially for condition accounts. A conceptual model makes it easier to identify the drivers of change and value in an ecosystem, the best ways to monitor the ecosystem, and knowledge gaps that need to be addressed. Conceptual models are effective tools for communicating restoration and accounting projects to stakeholders in an easily digestible way. Guidance on preparing ecosystem descriptions can come from other ecosystem accounts of the same ecosystem types, the **global ecosystem typology** for corresponding ecosystem functional groups, and other assessments, such as Red List of Ecosystems assessments[53]. Conceptual models are ecosystem specific, and tailored to the situation or area of interest, but can be modified from examples found in previous studies, such as the Red List of Ecosystems assessments, and in other ecosystem assessments and accounts[54].

Condition can be measured using a range of field-collected data and remotely-sensed data. For blue carbon ecosystem types dominated by woody vegetation – supratidal swamp forest, mangrove forests, Tectocornia shrublands – a range of remotely sensed variables, including vegetation metrics (e.g. vegetation greenness (NDVI), canopy height, age, cover, density, primary productivity) provide consistent broad scale measures of condition and are standard at a national or regional scale. In addition to the vegetation condition variables, incorporating indictors of overall ecosystem health and functioning will be important for an assessment of overall ecosystem condition. These indicators include presence and abundance of the suite of species (through direct observation, use of national and local datasets including Atlas of Living Australia), important species (e.g. First Nations' cultural importance, economic or social

[53] IUCN-CEM, et al. (2022). Red list of ecosystems. IUCN. https://iucnrle.org/

[54] Carnell, P.E., et al. (2022). Blue carbon drawdown by restored mangrove forests improves with age. Journal of Environmental Management 306:114301. https://doi.org/10.1016/j.jenvman.2021.114301

importance for example for recreational fishing, keystone species, and threatened species of concern), community assemblage measures (e.g. relative abundance of species), and biotic and abiotic processes. For example, in coastal wetlands, this would include changes in soil and water quality (e.g. water content, salinity, pH, turbidity, temperature, nutrients), and abiotic processes (e.g. flood and tidal regimes) and biotic processes (food web dynamics and abundance of key species).

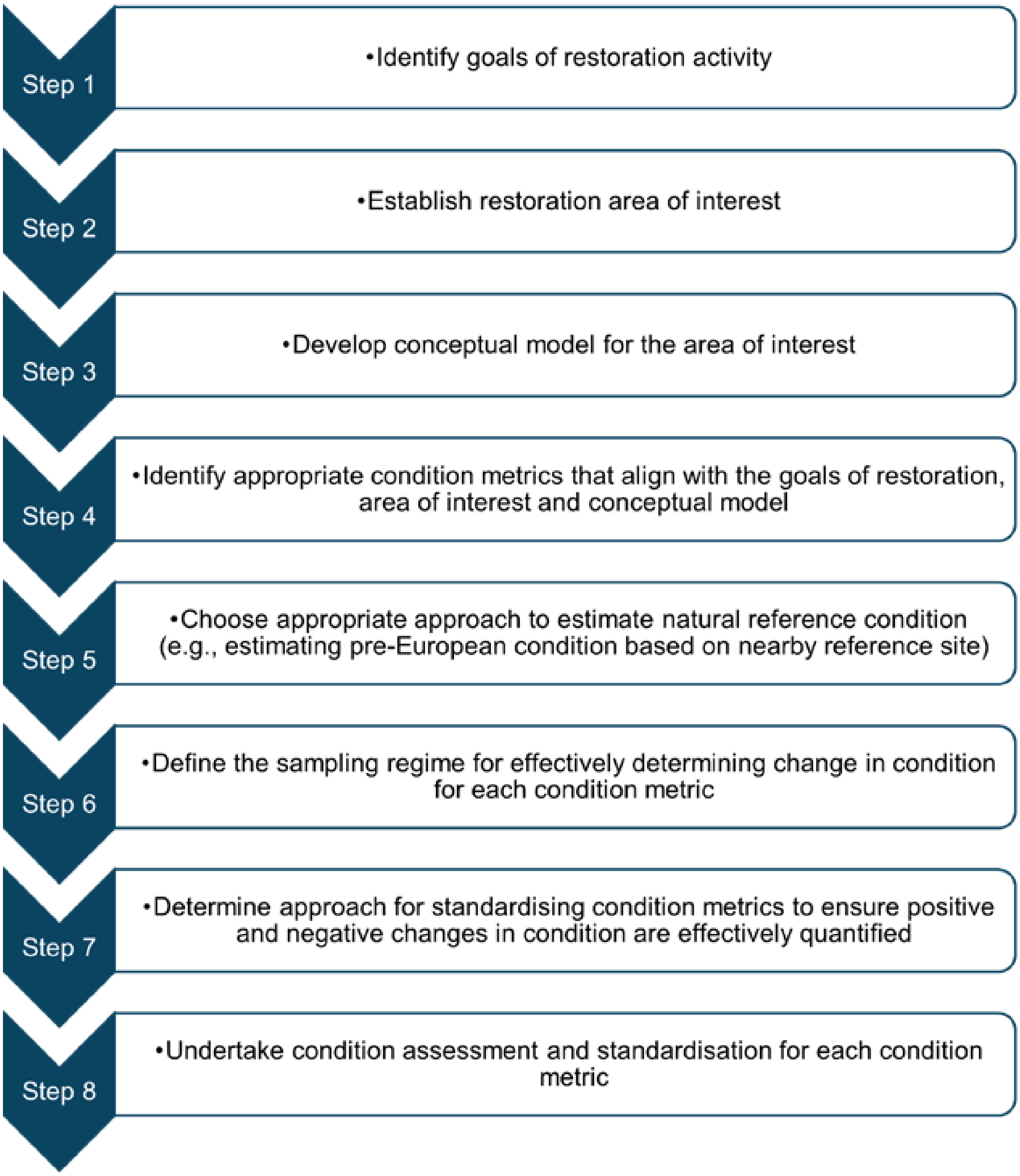

Figure 6.1: Flowchart to assess condition

Methods

Depending on the purpose of the restoration project and the ecosystems included, the components to measure will differ and should therefore be tailored to each restoration project utilising the framework and process described in Figure 6.1. Practitioners and experts in condition assessment should be engaged to do this.

Step 1: Identify goals of restoration activity

See Section 6.3 Ecosystem extent.

Step 2: Establish restoration area of interest

See Section 6.3 Ecosystem extent.

Step 3: Develop conceptual model for the site/ ecosystems

The first step to developing the list of variables to measure for the condition account, is to develop a conceptual model of the site, linking the ecosystems in consideration (see Figure 6.2 and conceptual models in Section 6.3) and combination of ecosystems as in Sievers et al.[55]). This should be done from the standpoint of the current ecosystems, threats, and values at the site. The conceptual model should also be done in a way as to consider how the restoration actions are predicted to influence the site, ecosystems, and values. Linking the conceptual model to the potential benefits of restoration is also important, because if the restoration activities are expected to benefit species of importance for recreational fisheries, it is also likely benefiting fish diversity and abundance more broadly. Thus, fish monitoring data should be included in both the condition (fish diversity) and services (nursery service or fisheries biomass) accounts.

Step 4: Identify appropriate condition metrics that align with the goals of restoration, area of interest and conceptual model

Based on the conceptual model, the user of the guide should choose condition variables to measure, that are best suited to track changes in the ecosystem in response to the threats being mitigated, such as species and components of the ecosystem that this is likely to impact. For example,

Figure 6.2 demonstrates how artificial seawalls reduce the hydroperiod and changes the salinity and pH to negatively impact mangrove trees. This impact on the mangrove trees in turn will impact the fish and crustacean species, which impacts the recreational fishing in the region. In this example, hydroperiod, salinity and pH would be the key physical and chemical variables to measure. Mangrove composition and structure (species present and biomass) are then critical compositional components to measure, as well as the fish and crustaceans that represent ‘end state’ indicators of condition directly relevant to stakeholder activities within the system.

Methods for monitoring condition variables can be similar across some habitats (e.g. for supratidal forests and mangroves) or quite different (e.g. saltmarsh compared to seagrass) between blue-carbon ecosystem types. This is due to differences in these ecosystem dynamics, but also the logistics and practicalities of remote-sensing and field approaches. Here recommendations have been split for approach by condition variable type and identified which ecosystem type they refer to for the lower cost (predominantly the remotely sensed approach Table 2.1) and the higher cost (field approach) (Table 5.1). While some low-cost approaches would be recommended as suitable compared to the high-cost approach (e.g. Normalised Difference Vegetation Index, forest age, above-ground biomass), other variables measured through the low-cost approach have their limitations (e.g. Hydroperiod can only be observed in unvegetated areas using spatial

[55] Sievers, M., et al. (2020). Integrating outcomes of IUCN red list of ecosystems assessments for connected coastal wetlands. Ecological Indicators. 116:106489. https://doi.org/10.1016/j.ecolind.2020.106489

methods, see these limitations in Trinity and Hunter case studies[57,58]). Therefore, combining a mixture of high and low-cost approaches is recommended depending on your project needs and the habitats found within the restoration site. For example, if the purpose of a restoration project is to increase habitat for migratory birds, the low-cost techniques for structural state, macroinvertebrate (bird diet) species richness through eDNA could be combined with high-cost technique for hydroperiod, salinity and bird abundance monitoring. Ideally, including variables from each of the SEEA Ecosystem Condition Typology (Physical state [A1], Chemical state [A2], Compositional state [B1], Structural state [B2], Functional state [B3], Landscape and seascape characteristics [C1]) are recommended to create a more complete condition account (see some examples within Table 2.1 and Table 5.1).

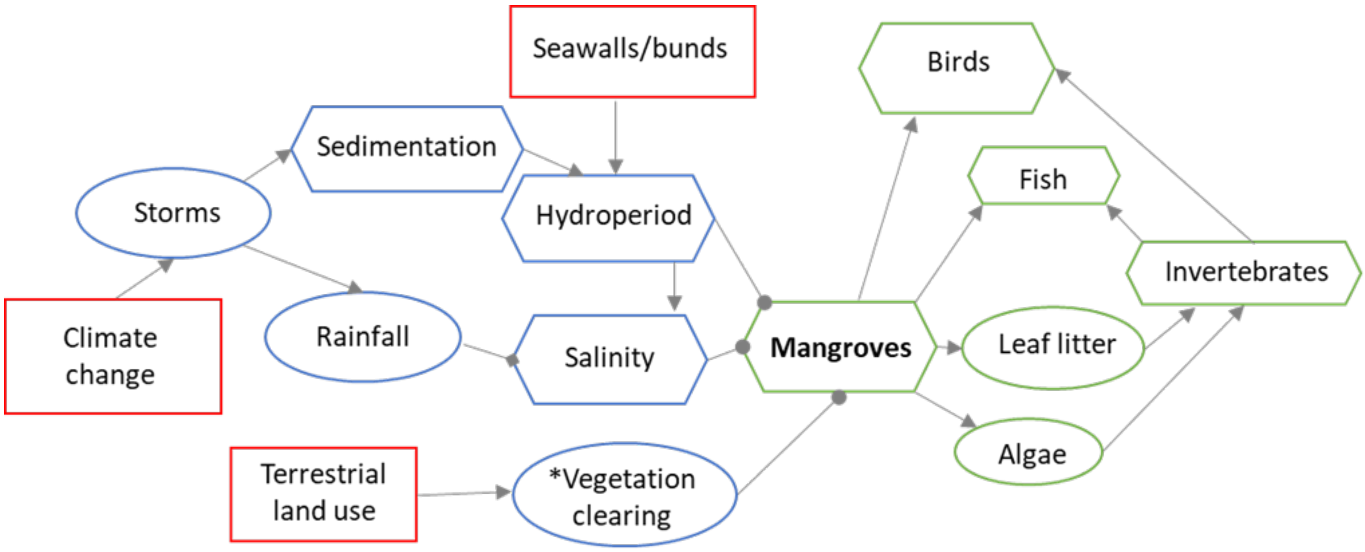

Figure 6.2: An example of a simplified conceptual model of the threats and the ecological processes relevant to mapping mangrove condition updated from Lee et al.[56] Red boxes indicate threats, blue ovals represent the abiotic processes, blue hexagons represent the abiotic environment, and the green hexagon represents the biotic components of the mangrove ecosystem, the ovals represent processes. Pointed arrowheads indicate positive effects, rounded arrowheads indicate negative effects, and diamond arrowheads indicate context-dependent effects. *denotes clearing for port or airport construction only in Australian context.

[56] Lee, C.K.F., et al. (2021). Mapping the extent of mangrove ecosystem degradation by integrating an ecological conceptual model with satellite data. Remote Sensing. 13(11):2047. https://doi.org/10.3390/rs13112047

[57] Glamore, W., et al. (2023). Accounting for benefits from coastal restoration: a case study from the Hunter River. Report to DCCEEW.

[58] Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW.

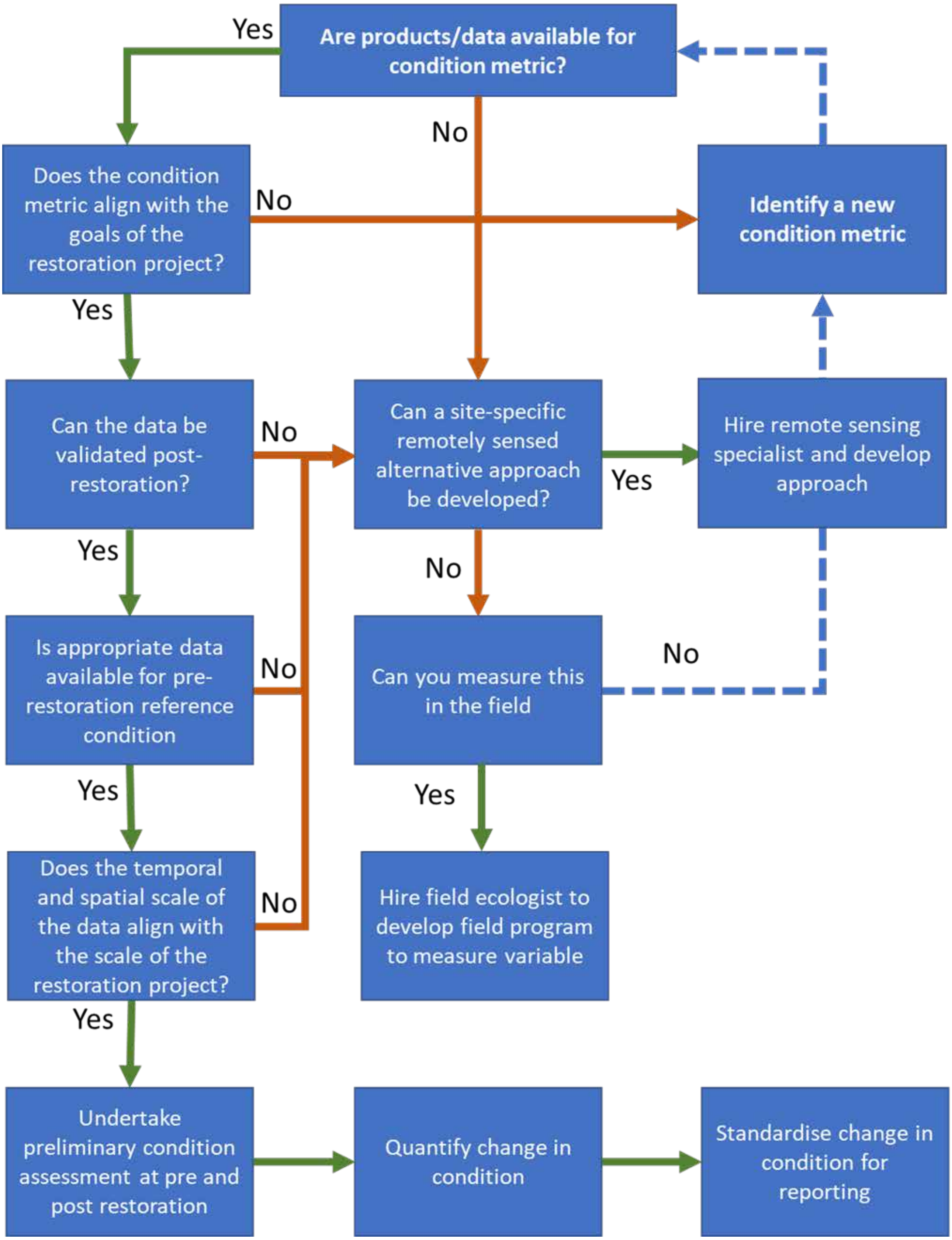

Figure 6.3: **Process for choosing approach to measure condition variables/metrics.**

Table 6.1: Examples of common condition variables for blue carbon ecosystems, lower cost, predominantly remotely sensed approach, based on Table 5.2 and 5.7 from United Nations et al. 2021[59] using the Ecosystem Condition Typology. Underlines indicate those applied in case studies.

Variable type	Intertidal vegetated (supratidal forest, saltmarsh, mangroves)	Intertidal seagrass or mudflats	Subtidal seagrass or unvegetated
Physical state (A1)	Elevation (Lidar)	Hydroperiod Water clarity (turbidity) Temperature Nutrients	Water clarity (turbidity) Temperature Nutrients
Chemical state (A2)		Nitrogen concentration	Nitrogen concentration
Compositional state; biodiversity (B1)	Bird species richness (modelling, eDNA, acoustic) Mammal species richness (modelling, eDNA) Reptile and amphibian species richness (modelling, eDNA, acoustic)	Fish species richness (modelling, eDNA) Macroinvertebrate species (modelling, eDNA) Shorebird species (modelling, eDNA, acoustic)	Fish species richness (modelling, eDNA) Crustacean species (modelling, eDNA)
<u>**Structural state (B2)**</u>	Normalised difference vegetation index Above-ground biomass Woody cover fraction[60] Tree height Canopy cover Forest age Age structure	Normalised difference vegetation index Seagrass cover Meadow age Fish biomass	
<u>**Functional state (B3)**</u>	Vegetation greenness Productivity	Productivity Top-predator populations (modelling, eDNA)	
<u>**Landscape and seascape characteristics (C1)**</u>	Patchiness/fragmentation Connectivity index	Patchiness/fragmentation Connectvitiy index	

[59] United Nations et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

[60] Liao, Z., et al. (2020). Woody vegetation cover, height and biomass at 25-m resolution across Australia derived from multiple site, airborne and satellite observations. International Journal of Applied Earth Observation and Geoinformation. 93:102209. https://doi.org/10.1016/j.jag.2020.102209

Table 6.2: As above, examples of common condition variables for blue carbon ecosystems, higher cost, field approach[61].

Variable type	Intertidal vegetated (supratidal forest, saltmarsh, mangroves)	Intertidal seagrass or mudflats	Subtidal seagrass or unvegetated
Physical state (A1)	Hydroperiod Sedimentation regime Elevation Temperature	Hydroperiod Sedimentation regime	Water clarity (turbidity)
Chemical state (A2)	Salinity Nitrogen pH Redox Sulfide Other nutrients Heavy metals	Salinity regime Nitrogen concentration pH Redox Sulfide Other nutrients Heavy metals	Nitrogen concentration pH Redox Sulfide Other nutrients Heavy metals
Compositional state; biodiversity (B1)	Bird species richness/key species abundance Mammal species richness Reptile and amphibian species richness Genetic diversity Ratio of native species	Plant species richness Bird species richness Macro-invertebrate spp. richness Genetic diversity Ratio of native species	Fish species richness Crustacean species richness Genetic diversity Ratio of native species
Structural state (B2)	Above-ground biomass Woody cover fraction Dominant tree height Canopy cover Forest age Age structure Reproduction/recruitment Leaf litter # pneumatophores Leaf litter	Above-ground biomass Plant height Plant cover % un-vegetated Marsh age Reproduction/recruitment	Seagrass length Seagrass cover Epiphyte cover Meadow age Reproduction/recruitment

[61] based on Table 5.2 and 5.7 from United Nations et al. 2021; https://seea.un.org/sites/seea.un.org/files/documents/EA/seea_ea_white_cover_final.pdf

Table 6.2: cont.

Variable type	Intertidal vegetated (supratidal forest, saltmarsh, mangroves)	Intertidal seagrass or mudflats	Subtidal seagrass or unvegetated
Functional state (B3)	Productivity Decomposition Water stress index Microbial diversity	Productivity Decomposition Microbial diversity	Productivity Top-predator populations Microbial diversity
Landscape and seascape characteristics (C1)	Patchiness/fragmentation Connectivity index	Patchiness/fragmentation Connectivity index	Patchiness/fragmentation Connectivity index

Criteria for selection of condition variables and data types: accuracy, precision, sensitivity and responsiveness* (read in conjunction with Table 6.3).

Once you have set your goals and decided on the condition variables you'd like to measure, you also need to figure out how you will measure these, and which will be able to meaningfully detect change. What you are looking for is data that can make your condition variable (or a combination of condition variables) into a condition 'index'[62]. There are criteria that need to be considered to do this well and they include precision, accuracy, and responsiveness of the data.

- **Precision** – is the precision level sufficient for measuring change? Do you have sufficient replication (e.g. Guillera-Arroita and Lahoz-Monfort[63])?
- **Accuracy** – can it be verified (e.g. consider source, metadata)? Does it need to be georeferenced? Do you need to use a combination of different data sources and integration processes to improve accuracy?
- **Responsiveness** – will the variable change due to the restoration action? At what spatial scale, magnitude, and in what timeframe is the variable likely to respond? For example, will species richness tell you enough about a change in condition following a restoration? For example, you may actually get no change in species richness, but a significant increase in the abundance of species. If your goal is to look at improvement in bird populations associated with saltmarsh restoration, bird species richness would then be a poor variable, and abundance of key species would be a better variable. Shannon's Index can also be an improved metric of bird biodiversity as it indicates if bird communities are dominated by few species or an even mixture. However, given that it may take time for wetland restoration to impact bird diversity or abundance, you could also survey the diversity and abundance of the macroinvertebrates. This is an indicator of condition in its own right, but may also include species that the target bird species feed on, and indicate if the habitat has likely been improved for bird

[62] Burgass, M. J., et al. (2017). Navigating uncertainty in environmental composite indicators, Ecological Indicators 75:268-278. https://doi.org/10.1016/j.ecolind.2016.12.034

[63] Guillera-Arroita, G., & Lahoz-Monfort, J.J. (2012). Designing studies to detect differences in species occupancy: power analysis under imperfect detection. Methods in Ecology and Evolution. 3:860-869. https://doi.org/10.1111/j.2041-210X.2012.00225.x

foraging. Macroinvertebrates are likely to respond on a quicker time-scale compared to bird populations, and measurements of both would provide complimentary suite of condition variables.

- **Will your data require treatment** to adequately measure change? E.g. means, medians, conversion to a ratio[64], integration into a control chart[65]. For example, in the case studies at Hunter and Trinity Inlet associated with this guide, identifying connectivity between ecosystems at the land/seascape scale used remote sensing data where pixels at the edges of differing ecosystems were used to provide a ratio of connectedness which was then reported as a scaled condition indicator between 0 and 1.

Step 5: Choose appropriate approach to estimate natural reference condition (e.g. estimating pre-European condition based on nearby reference site)

Condition accounts comprise measurements of several ecosystem-specific variables – for example salinity, tree height and bird diversity. Measurements of variables are converted into indicators, by normalising variables values to a 0-1 scale. This is done by comparing ecosystem specific variables from the area of interest with variables from a reference state that represents the 'natural state' of the ecosystem; in Australia this is defined as pre-European colonisation. This is different to the restoration goal, which may set a lower bar of condition given inherent changes in biota since European colonisation. The reference state is an exemplar of high condition (e.g. a value of 1); at the other end (e.g. a value of 0) is where the ecosystem has reached such a degraded state that it would be considered collapsed or transformed into a different ecosystem (e.g. from mangrove to mudflat when tree-cover goes below 0); the measurement variable for the area of interest is placed within this scale of high and zero condition.

In the following sections, the guide will give an example for how to select variables, identify appropriate datasets, and specify reference values for high and low condition to allow the development of indicators of condition. Condition indicators can be converted into a single condition index; however, this requires thoughtful weighting of the different indicators[66], and is not required, and, in many cases, not recommended.

A reference level is the value of a variable at the reference state, against which it is meaningful to compare past, present or future measured values of the variable. The difference between the value of a variable and its reference level represents the distance from the reference condition.

Following the steps outlined below, the value of the reference level is used to re-scale a variable to derive an individual condition indicator. Reference levels are defined in a structured and consistent manner across different variables within an ecosystem type, and for the same variable across different ecosystem types. This ensures that the derived indicators are comparable, and that their aggregation is biophysically meaningful.

Reference levels are usually set with high and low levels reflecting the limits or endpoints of the range of a condition variable that can be used in re-scaling. For example, the high condition reference level may refer to a natural state and the low condition reference level may refer to a degraded state where ecosystem processes are below a threshold for maintaining function (such as would be the case with ecosystem collapse[67]). One of the reference levels can often be replaced by the natural zero value of the variable, for example zero abundance (local extinction) for a species, or

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[64] Glamore, W., et al. (2023). Accounting for benefits from coastal restoration: a case study from the Hunter River. Report to DCCEEW.

[65] Carey, J., et al. (2017). Report on Condition of Yaringa Marine National Park - 2002 to 2013. Parks Victoria Technical Series No. 112. Parks Victoria, Melbourne.

[66] Burgass, M. J., Halpern, B. S, Nicholson, E. M., & Milner-Gulland, E. J. (2017). Navigating uncertainty in environmental composite indicators, Ecological Indicators 75:268-278. https://doi.org/10.1016/j.ecolind.2016.12.034

[67] Keith, D.A., et al. (2013). Scientific foundations for an IUCN Red List of Ecosystems. PLoS ONE 8(5): e62111. https://doi.org/10.1371/journal.pone.0062111

the lack of a specific pollutant. Reference levels applied to the same variables are likely to differ for different ecosystem types. For example, using the normalised difference vegetation index (NDVI) to indicate the variable of primary productivity will require reference levels for different ecosystems or ecosystem types (e.g. different mangrove species and structural communities, saltmarsh community species compositions).

Note that there are no 'default' natural state values that can be used across blue carbon ecosystems, because 'natural state' in one ecosystem may have indicators at opposite ends of the spectrum. For example, high nutrient and microalgae concentrations are typically associated with 'poor' condition, but for some systems that is representative of their 'natural' state[68].

Restoration goals: the desired outcomes of restoration at the site. This can include a general state, and targets for the specific indicators, and includes timeframes, given time lags in restoration outcomes. This is important for designing a monitoring strategy with the power to detect change, so you can know if you have achieved your goal (links with below, monitoring strategy).

One of the complexities with the reference state being defined as pre-colonisation, is that for many condition variables in coastal wetlands there may not exist historical data for which to set reference levels for these variables. This means users of the guide must use other approaches to estimating this reference state and choosing reference levels. Methods 1 - 4 should be considered first, before methods 5 - 7. It is important to be explicit in the approach you have used to set the reference condition state and levels. Further detail on these methods can be found in the UN SEEA-EA guidelines[69].

- **Method 1. Reference sites**: If pristine or minimally-disturbed sites are available, they can be used to determine a reliable measure of the mean and statistical distribution of condition variables. Given the complexities associated with measuring the environment and species at one site at one time, and comparisons to other sites, it is recommended to develop an appropriate experimental design based on the range of established literature (e.g. MBACI; Downes et al.[70], Keough and Quinn[71], Underwood[72] etc.), where your experimental treatment is the introduction of restoration actions. This will involve deciding on the number of before surveys, and the number of sites within the restoration area and sites within reference areas. This is because you can't reliably compare a survey of birds in winter before restoration to a survey of birds in summer after restoration, or that you may get inherent site differences in birds, unrelated to restoration action. Surveying reference sites is also important because they too may change over time due to other factors (e.g. climate), and thus users of the guide need to make sure that the changes that are detected at the restoration site are due to the restoration action rather than broader processes.

- **Method 2. Modelled reference conditions**: These can be based on predictive empirical models and can be used to infer conditions in absence of human disturbance where representative reference sites are not available. A weakness is that models are only as good as the data underpinning them and may oversimplify real-world conditions. Typically, these approaches have high levels of uncertainty, which makes assessing any change more difficult.

[68] Scanes, P., et al. (2007). Evaluation of the utility of water quality based indicators of estuarine lagoon condition in NSW, Australia. Estuarine, Coastal and Shelf Science, 74(1-2): 306-319. https://doi.org/10.1016/j.ecss.2007.04.021

[69] United Nations et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

[70] Downes, B. J., et al. (2002). Monitoring ecological impacts: concepts and practice in flowing waters. Cambridge, England: Cambridge University Press.

[71] Quinn, G. & Keough, K. (2002). Experimental Design and Data Analysis for Biologists. Cambridge University Press.

[72] Underwood, A. J. (1996). Experiments in Ecology. Cambridge University Press.

- **Method 3. Statistical approaches based on ambient distributions**: Least-disturbed conditions or best-attainable conditions can be estimated by observing the range of values from current ecosystem monitoring and by selecting a reference condition, for instance based on the 5th percentile values as criterion or by assuming that the reference condition is equal to a state with the highest species richness. Possible drawbacks are the arbitrary nature of the reference condition, spatial inconsistencies caused by using current datasets, a strongly shifting baseline that is no longer representative of natural conditions, or a false sense of consistency.

- **Method 4. Historical observations and paleo-environmental data**: Historical observations refer to a description of a reference condition based on species collections in natural history museums, historical manuscripts and books that describe fauna and flora, photo archives, paintings, or other material that can be used to make inferences about the presence of species or the prevalence of certain conditions during a certain period in time. Paleo-environmental data can be used to reconstruct the physical-chemical environment, climate, vegetation and fauna of certain period in time using material that is buried in the soil. A weakness is that not all ecosystem condition variables can be easily inferred from historical data, and historical data are rarely available or sparse.

- **Method 5. Contemporary data**: This method uses contemporary data to describe a contemporary reference condition. For instance, the Living Planet Index uses species data collected in 1970 as a reference to assess changes. However, there are several disadvantages. The choice of year may be considered arbitrary, especially in the context of blue carbon ecosystems that had been heavily modified across Australia post-colonisation. The reliance on contemporary data in evaluating changes can result in a shifting baseline. Appropriate dates differ for different indicators and ecosystem types. If different baseline dates are used in different regions this creates inconsistencies.

- **Method 6. Prescribed Levels**: Prescribed levels of a set of ecosystem condition variables can be used to construct a bottom-up reference condition. Examples of these reference levels include zero values for emissions or pollutants, a specific number of species, established sustainability or threshold levels such as critical loads for eutrophication and acidification, and target levels in terms of legislated quality measures (air and water quality). Prescribed levels are, however, not available for all variables, may be subject to policy influence and change over time, and may not be consistently developed for all ecosystem types, variables, or countries.

- **Method 7. Expert opinion**: This can be done using a structured survey approach ('expert elicitation') and is commonly used in natural resource management where data availability (or confidence in quality) is poor. It usually consists of a narrative statement of expected reference condition. Although an expert´s opinion may be expressed semi-quantitatively, qualitative articulation is probably most common[73]. Several weaknesses are inherently associated with this approach. Therefore, caution should be exercised.

- **Method 8. Combination**: Many of the above approaches may be used either singly or in concert for establishing and/or cross-validating reference condition. In practice, it may not be possible to use a single method to describe or quantify reference levels of ecosystem condition variables under a reference condition, and different approaches may need to be used for different variables. This approach reduces the likelihood that any one method will bias condition assessments.

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[73] European Commission. (2003). Common implementation strategy for the water framework directive (2000/60/EC). Guidance Document No 10. Rivers and Lakes – Typology, Reference Conditions and Classification Systems. Produced by Working Group 2.3 – REFCOND. http://www.wrrl-info.de/docs/Guidance_doc_10_REFCOND_klein.pdf

Step 6: Define the sampling regime for effectively determining change in condition for each condition metric

Qualified experts can set an appropriate sampling frequency and replication level of variables that will allow change to be quantified and reported in a robust way. The frequency and replication can vary significantly depending on the variables of interest and the purpose of the restoration project. A simplified process is outlined in Table 6.2 to guide effort during different project phases.

Monitoring frequency should be chosen based on the available budget, and the components of interest. For example, initially you may want to determine if the restoration actions have improved the physical and chemical variables at the site to allow for establishment of blue carbon ecosystems. Therefore, you may conduct the physical and chemical monitoring more frequently initially, in case any additional restoration actions need to be undertaken to ensure these variables improve. Once this is the case, plant and animal colonisation of the sites is a process that can take years to decades, and thus monitoring of these variables could be done more infrequently to suit the timeline of the project.

Keep in mind changes to hydrology for example may result in ecosystems that are not present at the beginning of the restoration project and may not be a specific end goal of the project e.g. intended saltmarsh but not intended supratidal forest. Therefore, your condition monitoring should be robust enough to adequately sample expected changes and allow for unexpected changes to be integrated and quantified. Although this can seem outside the scope of the initial project, the time period of restoration projects where EEA demonstrates the value of change is decadal, and at this scale projects have demonstrated different benefits to those initially predicted (e.g. increases in nearby fish biomass[74]). Condition monitoring before restoration, the 'baseline', is the most important for a restoration project and should be at the highest standard affordable. This allows for future improvements in data sources (e.g. emerging spatial products[75]) to be applied with confidence. For instance, where spatial data is inadequate to differentiate ecosystems (e.g. waterbodies only, can't distinguish what they are due to tidal time of imagery), field methods should be used to ground truth these as a priority.

[74] Glamore, W., et al. (2023). Accounting for benefits from coastal restoration: a case study from the Hunter River. Report to DCCEEW.

[75] See for example Espy Earth, http://www.espyearth.space/

Table 6.3: Recommended sampling frequency and replication by project phase (read in conjunction with **Criteria for selection of condition variables and data types: accuracy, precision, sensitivity and responsiveness*** (read in conjunction with Table 6.2)).

Phase of project	Recommended sampling	Reasoning	Relative spend
Before restoration (baseline)	Highest level of spatial and temporal replication possible Highest number of variables possible Ground-truthing of remote sensing methods wherever possible Consider season if relevant for key variables	Provides evidence of change due to restoration effort Future benefits not considered/ expected in scope of project can be measured in future Futureproofs for improved tech	$$$
Post-restoration initial	Weeks to months	Focus on variables matched to key outcomes Focus on variables likely to rapidly change	$$
Post-restoration short-term	Frequent (weeks-months) - time since restoration - seasonal	Focus on variables matched to key outcomes Focus on variables likely to rapidly change	$ - $$
Post-restoration long-term	Annual-bi-decadal Key season(s)	Focus on variables matched to key outcomes	$ - $$
Post-restoration 'end' of project	Decadal May not need to be as extensive as pre-restoration if interim monitoring was thorough Highest level of replication possible Highest number of variables possible Ground-truthing of remote sensing methods wherever possible Consider season if relevant for key variables	Provides evidence of change due to restoration effort Future benefits not considered/ expected in scope of project can be measured in future. Futureproof for improved technology May be a long time for site to be monitored again, so likely to be a key in time series/become a reference site for future restoration projects	$$ - $$$

Step 7: Determine approach for standardising condition metrics to ensure positive and negative changes in condition are effectively quantified

It is important to consider how best to summarise the condition variable information to accurately depict the changes in condition that have occurred at the site. This is because condition information will come from multiple sites within a restoration area, which may be responding differently to the restoration actions. For example, you may measure salinity at several sites, and each of these can be benchmarked against the reference levels chosen to convert this into a mean condition indicator for the site (see below). However, if some locations have seen an improvement in salinity conditions, while others have seen negative change, then it may be important to highlight the areas that have shown differences in response, rather than summarising as a mean value which may show little to no change.

In order to provide a richer description of the change in condition due to restoration activities,the approach used in the Hunter River and East Trinity Inlet case studies was to report on both the mean, as well as increased and decreased condition values[76,77]. This is also important in the adaptive management framework for a restoration site, which might mean additional restoration works are required to improve condition in the areas that have not improved. This extra information is achieved, where information on condition is from remote-sensing approaches (e.g. plant biomass) which cover the entire site, rather than discreate measures at individual sites. This information on condition can then be reported in a similar fashion as extent accounts, as the area that has increased or decreased in condition. To put this information into an accounting table, we have devised an ecosystem condition change matrix (Table 6.4). This is the same layout as the ecosystem extent change matrix presented in the UN SEEA-EA guidelines[78], but in this context, shows the transition between condition categories for each ecosystem, and also allows for change between condition categories and ecosystems (e.g. from degraded pasture to high condition saltmarsh following restoration). We recommend completing the standard condition account tables first, and then seek to expand on these with the approaches described above where appropriate.

Step 8: Undertake condition assessment and standardisation for each condition metric

Ecosystem condition indicators are rescaled versions of ecosystem condition variables. They are derived when condition variables are scaled against reference levels determined with respect to the reference state. Data values for each variable are initially transformed to a common dimensionless scale, with the two endpoints of the scale (or a range along the scale) representing a high condition value (1 or 100 %) and a low condition value (0 or 0 %) for that variable. While in some cases the top values for a variable can reflect a high condition score, the opposite is also possible, i.e. bottom values for a variable can reflect a high condition score, for instance for variables that measure pollution levels. When this occurs, scaling against the reference state should be modified to reflect the variation in the endpoints.

The transformed data can then be converted to ecosystem indicators. The simplest conversion uses two reference levels to reflect a high or low condition score. In this case, the indicator is calculated by a linear transformation shown in the formula below.

$$I = (V - V_L) / (V_H - V_L)$$

where I is the score of the indicator, V is the value of the variable, V_H is the high condition value and V_L is the low condition value.

[76] Glamore, W., et al. (2023). Accounting for benefits from coastal restoration: a case study from the Hunter River. Report to DCCEEW.

[77] Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW.

[78] United Nations, et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

Applying a reference level converts that variable from being a measure of trends in ecosystem characteristics to an assessment of ecosystem condition in relation to reference end points (i.e. V_H and V_L). Such normalisation adds value in the interpretation of trends and is also required by any later aggregation steps, which need commensurate metrics measured on the same scale using common units[79]. A set of indicators for a condition account can include some common or global indicators in addition to indicators specific to an ecosystem type. Examples of indicators are presented in Table 6.1 and Table 6.2.

There are examples where the relationship between a condition variable isn't linear like the approach outlined above. As one example, the hydroperiod for saltmarsh requires some inundation, but too much will see it convert into mangroves (e.g. sea level rise), and too little will see it convert into supratidal swamp or other drier ecosystem type (e.g. levee wall installation). Therefore, more complex equations converting the condition variable into the condition indicator will be required.

Indicator scoring

Indicator score at each time step (i_t) is:

$$i_t = \frac{(V_t - V_l)}{|(V_h - V_l)|}$$

Where:

V_t = is indicator value at time t

V_l = lowest feasible indicator value

V_h = highest feasible indicator value

With this in mind, the change in indicator can be considered as:

$$\frac{di}{dt} = \left(\frac{(V_2 - V_l)}{|(V_h - V_l)|}\right) - \left(\frac{(V_1 - V_l)}{|(V_h - V_l)|}\right)$$

Where:

V_2 = is indicator value at post-restoration time

V_1 = is indicator value at pre-restoration time

This can be reduced to:

$$\frac{di}{dt} = \left(\frac{(V_2 - V_1)}{|(V_h - V_l)|}\right)$$

In some cases, there is not a suitable pre-restoration indicator value (i.e. V_1) for condition scoring. This can occur where there is a change in state over time (i.e. saltmarsh changing to mangrove), or suitable data is not available to quantify the indicator at the pre-condition time. In these instances, V_1 should be substituted with a suitable reference value (V_r).

Description of variables and examples

Assessing ecosystem condition for environmental accounts is dependent on the accuracy of ecosystem extent delineation and validity of condition indicators in characterising the observed variation in condition. Using relevant existing condition indicators and mapping products, where available, provides consistency between assessments and facilitates comparison of condition between projects. Other products will continue to become available as they are developed and improved. Critically, these products may not be suitable for all project-level ecosystem accounts or not 'fit for purpose' based on the value proposition and project objectives upon which condition is being accounted.

To provide sufficient rigour, other data sources may be required based on project-level knowledge and restoration specifications. These may include existing scientific literature, historical ancillary mapping on vegetation health, historical and/or recent spectral imagery, land use information (i.e. local council cadastral information, ABARES products), and structural data from active sensors such as Lidar and radar, with the resolution of data modified on the basis of whether sensors are

[79] Nardo, M., et al. (2005). Handbook on Constructing Composite Indicators: Methodology and User Guide. OECD Statistics Working Papers, No. 2005/03, OECD Publishing, Paris. https://doi.org/10.1787/533411815016

fixed space-borne, airborne or remotely piloted aircraft. When using alternative data sources for extracting information on reference condition or condition metrics, protocols and methods may be required to ensure both suitability for the project and comparability to other project-level restoration activities. Moreover, these alternative data sources require greater expertise for quantifying ecosystem condition metrics.

Physical State & Chemical State

Measurements of the physical state of an ecosystem and restoration site are likely to be important for many projects. One common method of blue carbon ecosystem restoration is bund wall removal, which results in reinstating the natural hydrological regime to a site. Therefore, measurements of the hydrology, salinity and pH of the site are crucial to understanding if the restoration actions have been successful, which will then allow for colonisation by seagrass, mangrove, saltmarsh, and supratidal forest species.

High-cost (estimate $100k per component for each accounting period)

Several monitoring manuals provide a detailed breakdown of various components of the physical and chemical (and other ecosystem variables) that can be measured in the field and appropriate methods to measure them[80]. For example, hydroperiod is usually measured by deploying water level loggers at the site[81] (**Figure 6.4**), while salinity and pH can be measured with loggers or utilising hand-held probes[82]. This can allow for a highly accurate measurements and allow rapid changes to be identified and how this varies across small spatial scales. The methods themselves are commonplace and well established, but any monitoring that involves fieldwork will be inherently more costly than remote approaches.

Figure 6.4: Researcher deploying water level loggers, and measuring salinity and pH in a coastal wetland.

[80] Teutli-Hernández, C., et al. (2021). Manual for the ecological restoration of mangroves in the Mesoamerican Reef System and the Wider Caribbean. Integrated Ridge-to-Reef Management of the Mesoamerican Reef Ecoregion Project – MAR2R, UNEP-Cartagena Convention, Mesoamerican Reef Fund. Guatemala City, Guatemala.

[81] Glamore, W., et al. (2021). Eco-hydrology as a driver for tidal restoration: Observations from a Ramsar wetland in eastern Australia. PloS ONE, 16(8): e0254701. https://doi.org/10.1371/journal.pone.0254701

[82] Gouveia, M.M., et al. (2023). The Importance of Soil Elevation and Hydroperiods in Salt Marsh Vegetation Zonation: A Case Study of Ria de Aveiro. Applied Sciences, 13(7):4605. https://doi.org/10.3390/app13074605

Low-cost

There are several remote sensing approaches to measuring physical and chemical properties of coastal waters. This includes hydroperiod[83], ocean temperatures, salinity, turbidity, pH, and nutrients. Importantly, the main consideration is that this can only be achieved with reasonable confidence in areas without vegetation (i.e. mudflats; not mangroves, saltmarsh, or supratidal forests)[84]. Given the role of coastal wetland vegetation in restoration efforts, this will then limit the locations within the restoration site for which users can retrieve and use this information, and then provide only an indicator. Therefore, high-cost field measures are recommended where financially possible, especially at the commencement of the project and after it is 'complete'.

Compositional State (Biodiversity)

This is the key component of biodiversity measures. Measurements of the species within the ecosystems being restored are often a crucial component to assessing the condition of an ecosystem following restoration and may often be linked to some of the desired outcomes of a project (e.g. an increased diversity of fish or birds).

High-cost (estimate $100k per component for each accounting period)

There are standard field measures of assessing species diversity and abundance across a range of habitat types, these methods usually require multiple people to visually sample or detect species at multiple sites and times (see saltmarsh example[85], Figure 6.5). Environmental consultants and professional scientists can create customised monitoring strategies depending on the range of blue-carbon systems and their spatial scales at the restoration site.

Figure 6.5: Researcher conducting bird surveys in a coastal wetland.

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[83] Lee, C.K.F., et al. (2021). Mapping the Extent of Mangrove Ecosystem Degradation by Integrating an Ecological Conceptual Model with Satellite Data. Remote Sensing, 13(11):2047. https://doi.org/10.3390/rs13112047

[84] Lee, C.K.F., et al. (2021). Mapping the Extent of Mangrove Ecosystem Degradation by Integrating an Ecological Conceptual Model with Satellite Data. Remote Sensing, 13(11):2047. https://doi.org/10.3390/rs13112047

[85] Gouveia, M.M., et al. (2023). The Importance of Soil Elevation and Hydroperiods in Salt Marsh Vegetation Zonation: A Case Study of Ria de Aveiro. Applied Sciences, 13(7):4605. https://doi.org/10.3390/app13074605

Low-cost (estimate $10-$50k for each accounting period measured, depending on approach)

There are many new and developing methods for more quickly assessing diversity and abundance of coastal wetland species. Baited Remote Underwater Video (BRUVS) and camera traps have increased in popularity to survey speces requiring less time from researchers in the field, but still require significant post-processing time. Further to this, there are attempts to develop automated processing of video or camera trap data, but this is still in development. There are also other emerging low-cost approaches to measuring biodiversity such as eDNA, isotopic niche analysis, and bioacoustic monitoring. However, some of these methods are still in their infancy (depending on species application and ecosystem) and may require further research and development before they can be incorporated into these projects at low cost.

In the future it may also be possible to estimate increases in species based on models from rigorous datasets (where corrected for habitat availability). This would be done in a similar way as is done for fish[86] and carbon[87], where based on substantial datasets around the country, it would then be possible to conservatively estimate increases in species diversity and abundance. However, this data does not currently exist and would need substantial development to implement.

Structural State

High-cost (estimate $100k per component for each accounting period)

Like compositional state, there are several monitoring manuals for measuring the structural state of blue carbon ecosystems through field-based measures (e.g. for mangroves[88], seagrass[89], or tidal marsh[90], see **Figure 6.6**). However, there have been a number of recent advancements that can streamline field-based approaches, such as using Uncrewed Aerial Vehicles (UAV's, see **Figure 6.6**) for measuring plant structure and biomass (using Structure from Motion; SfM), which can save on field costs by an estimated $50,000 per ha due to reduced sampling effort and provide detailed and accurate data[91]. Structural state indicators are often proxies for function e.g. fragmentation may affect larval dispersal or the ability of seagrass to provide refuge from predators.

Low-cost ($10-100k for multiple remote sensing components)

As demonstrated in the Hunter[92] and East Trinity Inlet[93] case studies, there are ways to utilise remote sensing approaches to identify structural changes in vegetation, providing a proxy for identifying condition change due to restoration activities.

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[86] Jänes, H. et al. (2020). Quantifying fisheries enhancement from coastal vegetated ecosystems. Ecosystem Services, 43:101105. https://doi.org/10.1016/j.ecoser.2020.101105

[87] Lovelock, C.E., et al. (2022). An Australian blue carbon method to estimate climate change mitigation benefits of coastal wetland restoration. Restoration Ecology, e13739. https://doi.org/10.1111/rec.13739

[88] Teutli-Hernández, C., et al. (2021). Manual for the ecological restoration of mangroves in the Mesoamerican Reef System and the Wider Caribbean. Integrated Ridge-to-Reef Management of the Mesoamerican Reef Ecoregion Project – MAR2R, UNEP-Cartagena Convention, Mesoamerican Reef Fund. Guatemala City, Guatemala.

[89] Gamble C., et al. (eds) (2021). Seagrass restoration handbook. Zoological Society of London, UK., London, UK.

[90] NYC Parks. (2018). Saltmarsh restoration monitoring guidelines. City of New York Parks and Recreation, Forestry, Horticulture, and Natural Resources, New York.

[91] Navarro, A., Young, M., Allan, B., Carnell, P., Macreadie, P., & Ierodiaconou, D. (2020). The application of Unmanned Aerial Vehicles (UAVs) to estimate above-ground biomass of mangrove ecosystems. Remote Sensing of Environment, 242, 111747. https://doi.org/10.1016/j.rse.2020.111747

[92] Glamore, W., et al. (2023). Accounting for benefits from coastal restoration: a case study from the Hunter River. Report to DCCEEW.

[93] Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW.

Where national products are available and appropriate for the project-level ecosystem accounts, some expertise in Geographic Information Systems (GIS) will be required. Most national products will be available in relevant spatial data formats that can be extracted from data portals using the project-level area of interest. As an example, when mapping mangrove ecosystem condition with Digital Earth Australia (DEA) Mangroves canopy cover, the following steps should be undertaken:

- Identify area of interest (latitude and longitude) and time period (pre-restoration, post-restoration years).
- Data for area of interest extracted out using DEA data portal Data loaded into GIS software (e.g. QGIS, ArcGIS).
- Mangrove total canopy cover (low, medium, dense) calculated using zonal statistics tool.
- Change in canopy cover represented as ecosystem condition change based on value of mature, resilient mangrove forests.

Where national products are not available or are not suitable for identifying project-level condition change, moderate to considerable GIS and remote sensing expertise will be required. Multiple approaches can be taken to provide sufficient rigour, and selection of a suitable approach will depend on the available GIS expertise. For example, extracting mangrove canopy cover from analysis-ready digital airborne imagery, assessment will require:

- Identifying and extracting by mask the restoration area of interest.
- Using extracted mangrove area polygons from ecosystem extent to identify pixels only relevant to mangrove vegetation.
- Calculating Foliage Projective Cover (FPC[94]).
- Establishing change in FPC as ecosystem condition change based on the value of intact, resilient mangrove forests.

These approaches are dependent on data suitability and GIS expertise and have varying degree of accuracy limitations.

Figure 6.6: A) Researchers measuring mangrove density and diameter. B) Researchers using drones to map vegetation structure in coastal wetlands.

[94] Asbridge E., et al. (2018). The extent of mangrove change and potential for recovery following severe Tropical Cyclone Yasi, Hinchinbrook Island, Queensland, Australia. Ecology and Evolution, 8:10416-10434. https://doi.org/10.1002/ece3.4485

Functional state

High-cost (estimate $100k per component for each accounting period)

Some common measure of structural state in coastal wetlands include productivity (plant biomass produced per area per year), decomposition (measure of organic material degradation over a period of time, see Figure 6.7), which is also linked to microbial diversity, and diversity of species within different functional groups (e.g. predators (shark), herbivores (turtles), producers (seagrass)).

Low-cost

By using remote-sensing approaches, proxies for ecosystem function can be generated. These include measures such as vegetation greenness (Normalised difference Vegetation Index), vegetation wetness (Normalised Difference Moisture Index), above-ground biomass, canopy cover, and vegetation age. It is important that these measures are used in the context of measures that may indicate functional state, with varying levels of confidence that can be ecosystem type specific.

Figure 6.7: Researcher measuring decomposition using a standardized teabag index method.

Landscape and seascape characteristics

High-cost (estimate $100k per component for each accounting period)

Connectivity and patch-scale dynamics can be important to the condition of a restoration site. If there is a lack of connectivity to nearby healthy blue carbon ecosystems, then recolonisation and recovery at the restoration site may not occur, or occur slowly. A higher-cost approach here would be to run connectivity models using data on characteristics of the water (e.g. tidal currents, water velocity), and characteristics of the plant or animal dispersing in the water (e.g. do the larval fish swim, or does the plant propagule float[95] etc.). Collecting these data will usually require extensive on-ground fieldwork.

Another metric, may be the patch size and connectivity within the restoration site itself. Patch number and size are known to influence species that inhabit those patches[96,97]. Here, the data obtained from the extent account can then be combined with knowledge of patch dynamics for the ecosystems and region in question. The higher cost approach here would be to map patch size through field measures such as drones and AUVs.

Low-cost

A lower cost approach to measuring connectivity is based upon average distances of dispersal from numerous studies, in combination with the remotely sensed extent data (e.g. connectivity analysis in case studies[98,99]). Similarly, patch-size metrics can be ascertained from remote sensing approaches. Where connectivity between ecosystem types and across landscapes for particular ecological processes and species dispersal is well understood, remotely sensed fragmentation metrics can be effective tools for approximating relative connectivity. Relatively simple GIS analysis can be applied to identify change in relative connectivity of an ecosystem type using a moving window analysis (see case studies). This analysis will be underpinned by level of confidence in ecosystem extent measures.

Key assumptions or limitations

Ecosystem condition is useful for identifying changes in a landscape caused by outside forces to implement timely and effective management responses. It is also a useful for quantifying positive or negative outcomes following an activity or management action. Inherently, condition is a value-based assessment that must be in reference to an aim or objective. An assessment of condition may not provide a complete overview of the state or trajectory of an ecosystem and relies on a presumption that the value-based objective is suitable for quantification through a condition assessment. Condition is typically an assessment where a metric is considered relative to a reference or baseline; however, selection of an appropriate baseline may be difficult, particularly on coastal ecosystems that have been heavily impacted since European occupation in Australia. With this in mind, it is important that ecosystem condition for environmental accounting avoids misrepresentation of condition by clearly outlining what the condition assessment indicates about the current ecosystem state and projected condition.

A limitation to the lower-cost remote sensing approaches is that the suggested methodologies may be limited by remote sensing data availability, and suitability for identifying change in condition due to restoration activities. This could be a lack of suitable data (e.g. cloud cover, no existing pre-restoration activity imagery, lack of time-step or season of interest), inability to detect meaningful change in condition with restoration activities,

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[95] Van der Stocken, T., et al. (2018). Global-scale dispersal and connectivity in mangroves. PNAS, 116(3):915-922. https://doi.org/10.1073/pnas.1812470116

[96] Smith, T.M., et al. (2010). Seagrass patch size affects fish responses to edges. Journal of Animal Ecology, 79: 275-281. https://doi.org/10.1111/j.1365-2656.2009.01605.x

[97] Macreadie, P.I., et al. (2009). Fish Responses to Experimental Fragmentation of Seagrass Habitat. Conservation Biology, 23: 644-652. https://doi.org/10.1111/j.1523-1739.2008.01130.x

[98] Glamore, W., et al. (2023). Accounting for benefits from coastal restoration: a case study from the Hunter River. Report to DCCEEW.

[99] Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW

or inherent ecosystem characteristics that can cause conflicting interpretations on the condition of different ecosystems. Where this is important, field data will need to be incorporated. For example, an increase in mangrove canopy cover after restoration activities suggests an increase in condition of mangrove vegetation (i.e. increase biomass, forest maturity, increased recruitment, increased resilience); however, this could be a response to other environmental drivers (e.g. sea-level rise), rather than an outcome of restoration activities. Expansion of mangrove area and increases in canopy cover in adjacent saltmarsh should also be considered in the context of the negative impacts on saltmarsh values, such as water bird habitat.

Finally, GIS and remote sensing expertise will be essential to condition accounts. Users with less expertise may only be able to use of data products that are less suitable, and this will limit the capacity to successfully delineate coastal ecosystems (e.g. waterbodies). The limitations of data and expertise will influence map accuracy and calculations of ecosystem condition. Given that GIS expertise is required in the extent accounts as well, it is suggested that a GIS expert complete both the extent and the relevant condition variables. They should work with ecologists and ecosystem experts to ensure appropriate selection of indicators that can be generated with appropriate level of confidence, reference levels and datasets, and interpretation of results.

Data sources

Remotely sensed data

Several freely available products can be used to map ecosystem extent. These include:

Existing products

- **Mangroves canopy cover**: DEA Mangroves (https://cmi.ga.gov.au/data-products/dea/634/dea-mangrove-canopy-cover-landsat).
- **Saltmarsh**: Australian Saltmarsh Map (https://www.saltmarshes.org/home). This product will be publicly available in 2023.
- **Supratidal Forests**: An Australia-wide product is currently in development by the authors. product will be publicly available in 2024.
- **Intertidal seagrass**: An Australia-wide product is currently in development. This product will be publicly available in 2023. Interim Intertidal seagrass data was sourced from an experimental earth observation-based product commissioned from the University of New South Wales by DCCEEW and supplied to the ABS.
- **Waterbodies**: DEA Land Cover (https://www.dea.ga.gov.au/products/dea-land-cover).
- **Mudflats**: Global Intertidal change (https://www.intertidal.app/).
- **'Other' ecosystems**: DEA Land Cover (https://www.dea.ga.gov.au/products/dea-land-cover).
- **Biomass**: for mangrove, supratidal swamp forest, pre-restoration environments can be determined using several products existing at global and national levels between 25-250 m spatial resolution with varying time-series available. These products are well documented for mangrove vegetation and are likely broadly suitable for supratidal swamp forest and pre-restoration vegetation. These include ESA CCI biomass maps, GEDI biomass maps, Mangrove biomass from SRTM measurements[100].
- Digital Earth Australia Land Cover, providing annual land cover information (> 100 land cover attributions) for Australia at 25 m resolution from 1988-2020[101,102,41].

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[100] Accessed at: https://climate.esa.int/en/projects/biomass/data/; https://gedi.umd.edu/data/products/; https://daac.ornl.gov/CMS/guides/CMS_Global_Map_Mangrove_Canopy.html

[101] Owers, C. J., et al. (2022). Operational continental-scale land cover mapping of Australia using the Open Data Cube. International Journal of Digital Earth, 15(1), 1715-1737. https://doi.org/10.1080/17538947.2022.2130461

[102] Accessed at: https://cmi.ga.gov.au/data-products/dea/607/dea-land-cover-landsat

- **Supratidal swamp forest**: Woody Cover Fraction (WCF), providing annual fraction of woodiness of vegetation for Australia at 25 m resolution from 1988 – 2020/19.

Data availability: requires processing to be useful

- In addition to national approach: historical aerial imagery, Nearmap aerial imagery, RPA footage (drone).
- 2014 canopy height model (CHM).
- Publicly available 2018 AGB derived from the European Space Agency's (ESA's) Climate Change Initiative Biomass project (CCI Biomass).

Data availability: requires field collection or processing

While remotely sensed data rely on field-data for training and validation/accuracy assessment, field data are also valuable in their own right for quantifying ecosystem condition. This can include 'fit-for-purpose' detailed surveys at the area of interest before and after restoration efforts. Baseline data can also be derived from existing sources, such as Atlas of Living Australia, state based data sources (such as those in South Australia or Victoria) or long-term monitoring programs. Such data are typically modelled, statistically to examine temporal trends, and spatially, to extrapolate to areas not surveyed (e.g. via species distribution models, and community assembly or diversity models).

- Atlas of living Australia.
- Birdlife Australia.
- EPA – water quality data.
- Department of Fisheries.

Local stakeholders will often have valuable field perspectives on historical patterns of ecosystem condition. These can be used to inform restoration targets or to validate the appropriateness of the data sources used elsewhere if they fail to capture patterns identified by stakeholders. While these perspectives are often not quantitative, they are key to ensuring the datasets selected elsewhere identify patterns of interest.

- Field naturalist groups – many specialist species groups that collect verified records.
- Rangers.
- Traditional Owners.

Uncertainties

Assessing condition of coastal wetlands can be a complex task, requiring trained experts to plan, action and analyse results to manage uncertainties and ensure interpretation of data is accurate. Choosing indicators that are sensitive enough to identify change also need to be interpreted within natural fluctuations such as seasonality, life history and scale. Multiple data sources can both reduce and increase uncertainty. Indexes such as control charts and ratios can be useful to manage these uncertainties.

Additionally, including measurements of the variability (Mean + Standard Error, percentiles etc.) in condition indicators can also be useful for understanding uncertainty in the measures you have recorded at a site. Conducting a power analysis can also be a useful way of understanding the ability you have to detect meaningful change based on the indicator and the sampling design you have employed at the site[103].

[103] Quinn, G. & Keough, K. (2002). Experimental Design and Data Analysis for Biologists. Cambridge University Press.

6.3 Supplementary material

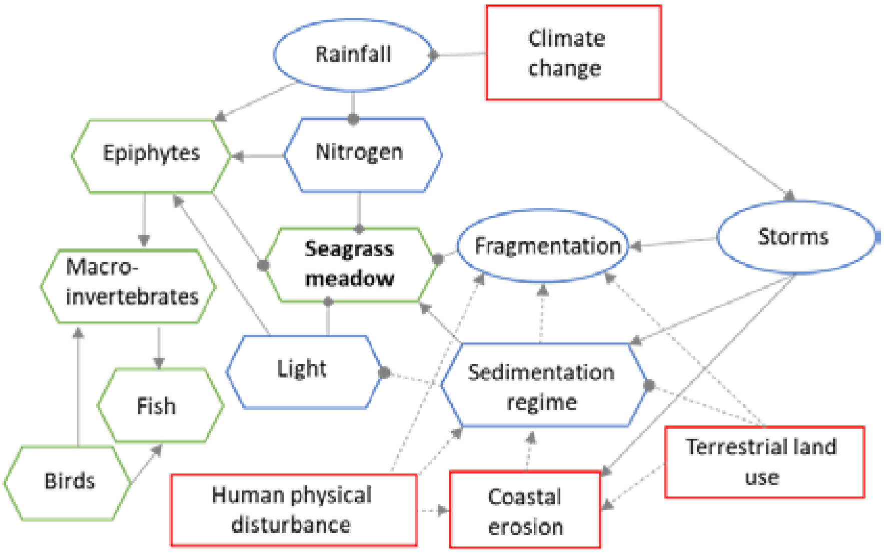

Figure 6.8: **An example of a simplified conceptual model of the threats and the ecological processes relevant to assessing seagrass condition, adapted from Carnell et al. (2022)[104]**. Red boxes indicate threats, blue ovals represent the abiotic processes, blue hexagons represent the abiotic environment, and the green hexagon represents the biotic components of the seagrass ecosystem. Pointed arrowheads indicate positive effects, rounded arrowheads indicate negative effects, and diamond arrowheads indicate context-dependent effects. Unbroken linking line indicates known affect, broken link indicates possible affect.

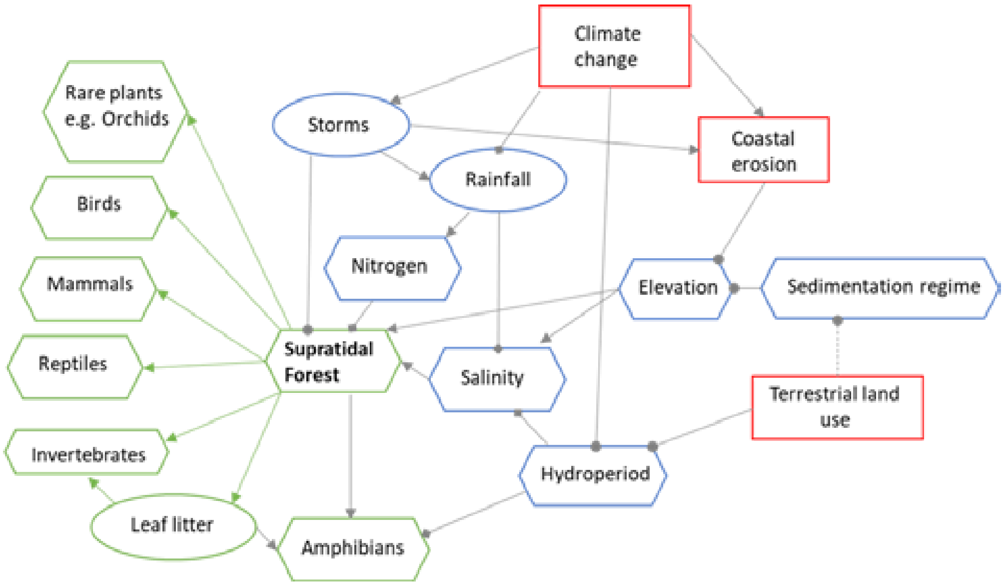

Figure 6.9: **An example of a simplified conceptual model of the threats and the ecological processes relevant to assessing supratidal forest condition**. Red boxes indicate threats, blue ovals represent the abiotic processes, blue hexagons represent the abiotic environment, and the green hexagon represents the biotic components of the supratidal forest ecosystem. Pointed arrowheads indicate positive effects, rounded arrowheads indicate negative effects, and diamond arrowheads indicate context-dependent effects. Unbroken linking line indicates known affect, broken link indicates possible affect.

[104] Carnell, P., et al. (2022). Prioritising the restoration of marine and coastal ecosystems using ecosystem accounting. https://doi.org/10.21203/rs.3.rs-1617940/v1

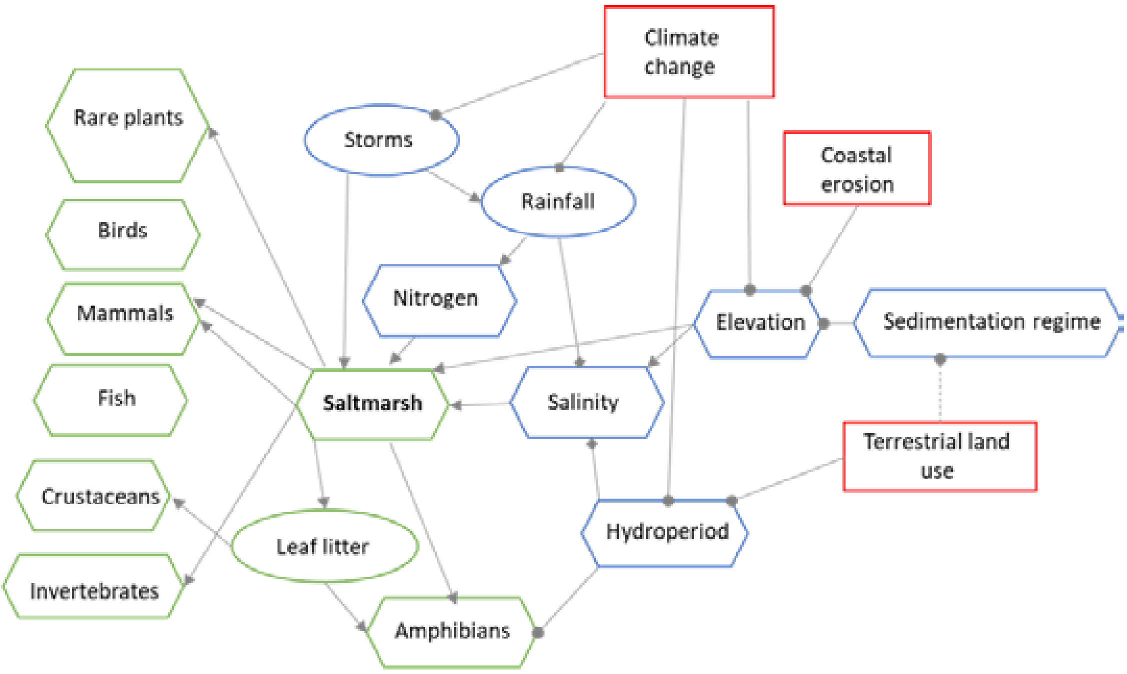

Figure 6.10: An example of a simplified conceptual model of the threats and the ecological processes relevant to assessing saltmarsh condition. Red boxes indicate threats, blue ovals represent the abiotic processes, blue hexagons represent the abiotic environment, and the green hexagon represents the biotic components of the saltmarsh ecosystem. Pointed arrowheads indicate positive effects, rounded arrowheads indicate negative effects, and diamond arrowheads indicate context-dependent effects. Unbroken linking line indicates known affect, broken link indicates possible affect.

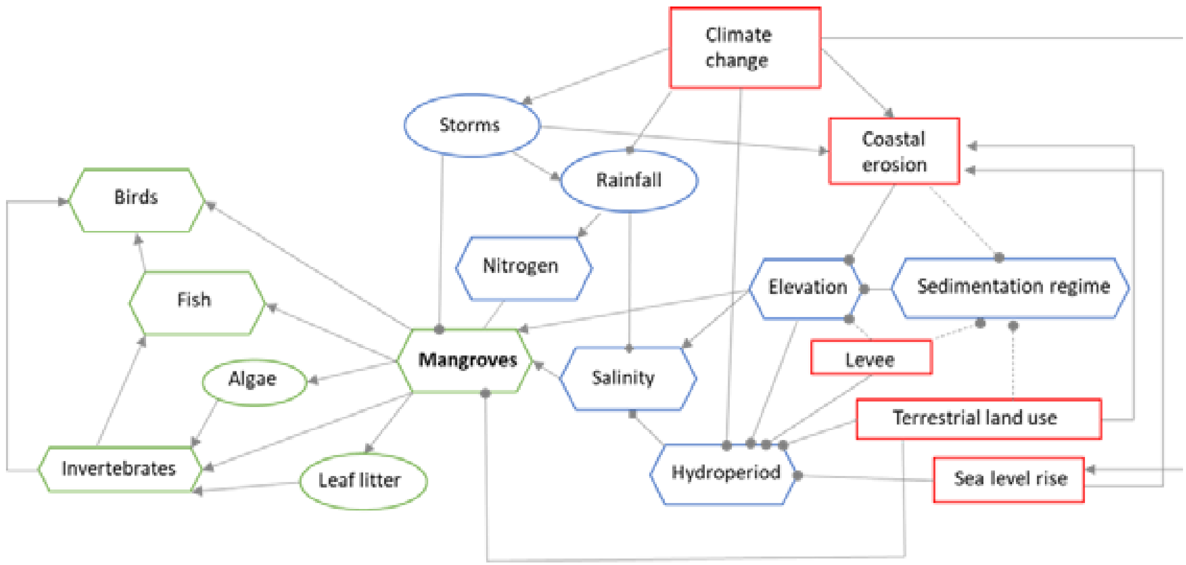

Figure 6.11: An example of a simplified conceptual model of the threats and the ecological processes relevant to assessing mangrove condition. Red boxes indicate threats, blue ovals represent the abiotic processes, blue hexagons represent the abiotic environment, and the green hexagon represents the biotic components of the seagrass ecosystem. Pointed arrowheads indicate positive effects, rounded arrowheads indicate negative effects, and diamond arrowheads indicate context-dependent effects. Unbroken linking line indicates known affect, broken link indicates possible affect.

Table 6.4: Example of an Ecosystem condition change matrix using simulated data. Based on the ecosystem type change matrix in the SEEA-EA guidelines.

			Ecosystem condition - closing (ha)																
			Saltmarsh				Mangroves				Mudflat				Pasture				
			High	Medium	Low	Collapsed	High	Medium	Low	Collapsed	High	Medium	Low	Collapsed	High	Medium	Low	Collapsed	Opening condition (ha)
Ecosystem condition - opening (ha)	Saltmarsh	High	2	5															7
		Medium		3															3
		Low			10														10
		Collapsed	10	5	2	5													22
	Mangroves	High					5												5
		Medium					20	10											30
		Low						10											10
		Collapsed						5											5
	Mudflat	High																	0
		Medium									5								5
		Low					5	10				5							20
		Collapsed										5							5
	Pasture	High													2				2
		Medium														3			3
		Low	50	20			5	5				5					5		90
		Collapsed																	0
		Opening condition (ha)	62	33	12	5	35	40	0	0	5	15	0	0	2	3	5	0	

Biodiversity accounts

While direct and indirect indicators of biodiversity such as species richness or abundance may be measured within condition accounts, practitioners may want to assess biodiversity-related data in a separate account. Since this guide already covers aspects of biodiversity in the previous condition section, a full guide on biodiversity accounts is not included. Instead, we provide a placeholder with some resources on the types of data and measurements that could be taken to form biodiversity accounts, including suggesting some future methods that could be used to quickly assess biodiversity and biodiversity-related measures for the purpose of economic accounting.

The concept of specific biodiversity accounts is relatively novel, having started in the late 1990s but expanding significantly after 2010, and many of these used a SEEA framework[105]. Adoption and consideration of biodiversity accounts in conservation has, however, been slow. This is possibly a result of difficulties standardising quantitative biodiversity accounting approaches across different habitats (e.g. terrestrial vs aquatic) which often require different metrics or expertise. Like condition accounts, broader restoration objectives are also relevant when designing biodiversity accounts. For example, broader biodiversity outcomes such as diversity and abundance may be less relevant than the presence of one endangered species. Ecological function may also be a desirable outcome, such as successful nesting of seabirds.

To date, most biodiversity accounts focus on three aspects of biodiversity: ecosystem extent, diversity, and abundance. While such an approach might be applicable to national accounts[106], on their own they are likely too coarse for project-level restoration. Extent of ecosystems is already collected as part of extent accounts, so should not be double-counted in biodiversity accounts. Abundance and diversity of species should be necessary minimums to build biodiversity accounts, but reducing ecosystems to these most basic biodiversity indicators oversimplifies ecosystem function, and may not be able to detect changes in ecosystem function that are relevant to restoration objectives (see above). We suggest that, using expert advice informed on the objectives of the restoration project, additional indicators of functional ecology are included within the biodiversity accounts.

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[105] Blanco-Zaitegi, G., Etxeberria, I.Á. and Moneva, J.M., 2022. Biodiversity accounting and reporting: A systematic literature review and bibliometric analysis. Journal of Cleaner Production, p.133677.

[106] Bogaart, P., Polman, E., Verweij, R. and van Swaay, C., 2020. The SEEA-EEA experimental biodiversity account for the Netherlands. Statistics Netherlands.

Future directions

As explained in the condition section, traditional on-ground assessments of biodiversity and biodiversity-related measurements that would be required to build biodiversity accounts can be costly. This is in part due to the time in the field that is required, and field costs scale in relation to the size of restoration sites. Analysis of data collected in the field can also be costly and time-intensive due to the nature of assessing biodiversity-related data such as video surveys often used in aquatic environments. One approach to reduce costs here could be the use of broader species distribution models[107], but like all models their use in these applications may be contentious given the underlying data need to be validated, possibly negating the cost-savings[108].

Testing and validating novel methods that reduce costs of building biodiversity accounts would be very beneficial to these sorts of applications. For example, eDNA has the potential to revolutionise diversity assessments, but on its own cannot provide important data on abundance of separate species. In addition, traditional biodiversity metrics exclude ecosystem function (fish populations in a restored site may be as abundant as in natural habitats, but these species may not interact in natural ways and be more vulnerable to perturbations[109]).

[107] Hein, L., Remme, R.P., Schenau, S., Bogaart, P.W., Lof, M.E. and Horlings, E., 2020. Ecosystem accounting in the Netherlands. Ecosystem Services, 44, p.101118.

[108] Araújo, M.B., Anderson, R.P., Márcia Barbosa, A., Beale, C.M., Dormann, C.F., Early, R., Garcia, R.A., Guisan, A., Maiorano, L., Naimi, B. and O'Hara, R.B., 2019. Standards for distribution models in biodiversity assessments. Science advances, 5(1), p.eaat4858.

[109] Maureaud, A., Hodapp, D., Van Denderen, P.D., Hillebrand, H., Gislason, H., Spaanheden Dencker, T., Beukhof, E. and Lindegren, M., 2019. Biodiversity–ecosystem functioning relationships in fish communities: biomass is related to evenness and the environment, not to species richness. Proceedings of the Royal Society B, 286(1906), p.20191189.

Ecosystem services - physical and monetary

Introduction to ecosystem services and EEA

In the ecosystem accounting framework, ecosystem services serve as the connecting concept between ecosystem assets and the production and consumption activity of businesses, households, and governments. The measurement of ecosystem services is thus central to describing a set of ecosystem accounts[85]. The measurement of ecosystem services is used for explaining the variety of contributions that ecosystems make to people and the economy. These contributions extend beyond marketed goods, such as timber and fish, and include services such as water purification, global climate regulation and recreation-related services. Commonly, these types of services are supplied to communities outside markets. The focus of accounting for ecosystem services is to provide a clear description of the range of these services, the spatial heterogeneity of their delivery, and the local to global beneficiaries of these services. This information can be compared between and connected to the different ecosystems that supply the services[86].

Importantly though, accounting for ecosystem services does not provide a complete assessment of the relationship between ecosystems and people. While the scope of ecosystem services is broad, there are a range of other benefits that are not captured, for example relational and intrinsic values. Nonetheless, a focus on ecosystem services does provide an important piece of information in describing our use of, and dependence on ecosystems. Further, together with information on the extent and condition of ecosystem assets, data about expenditure on environmental protection and resource management, and data on economic activity, a rich picture of the relationship can be portrayed.

The key concepts of the ecosystem accounting framework related to ecosystem services concern (i) the supply of ecosystem services to users; and (ii) the contribution of ecosystem services to benefits (i.e., the goods and services ultimately used and enjoyed by people and society). Ecosystem services are defined in the SEEA-EA as 'the contributions of ecosystems to the benefits that are used in economic and other activity'[87]. In this definition, 'use' incorporates direct physical consumption, passive enjoyment, and indirect receipt of services.

Further, ecosystem services encompass all forms of interaction between ecosystems and people including both those within the ecosystem and remote interactions. In ecosystem accounting, ecosystem services are recorded as flows between ecosystem assets and economic units; where economic units encompass the various institutional types included in the national accounts, such as businesses, governments, and households. Flows of ecosystem services are sometimes reflected in direct physical flows, such as when fish are removed from a marine ecosystem, but may also be reflected in the indirect receipt of ecosystem services, such as flood control services.

Following the cascade model describing flows of ecosystem services, the supply of an ecosystem service will be associated with an ecosystem structure or process or a combination of

[85] United Nations, et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

[86] United Nations, et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

[87] United Nations, et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

ecosystem structures and processes that reflect the biological, chemical and physical interactions among ecosystem components[88]. These processes and characteristics are observable and measurable but are not themselves flows of ecosystem services as defined in ecosystem accounting since this requires a connection to be made to users. This alignment between supply and use is a foundational accounting concept[89] and applies in both physical and monetary terms. The recording of ecosystem services will apply to total flows over an accounting period (e.g., one year) and thus an entry will reflect a total flow per unit of time.

The relationship between the supply of ecosystem services and the use of ecosystem services will not always be from one ecosystem asset to one economic unit or user. In some cases, ecosystem services will be supplied through a combination of ecosystem assets, for example flood control services involving a range of ecosystem types within a catchment. In other cases, one ecosystem service will be used by different economic units. For example, air filtration services will contribute to benefits used by both households and businesses. In some cases, the ecosystem services will be an indirect contribution to benefits, for example, where the nursery population services supplied by seagrass meadows are an input to the supply of fish biomass provisioning services, which in turn contribute to the benefit of marketed fish. In this case, the nursery population service is treated as intermediate while the biomass provisioning service is final.

Monetary valuation

Benefits are the goods and services that are used and enjoyed by people and society. As applied in ecosystem accounting, a benefit will reflect a gain or positive contribution to well-being arising from the use of ecosystem services. As we can measure ecosystem services in physical terms, so too can we measure them in monetary terms, which allows us to understand the total scale of value produced by a restoration activity, as well as understand the relative scale of the many ecosystem services that a project produces.

While measuring monetary benefits for ecosystem accounting provides us with an understanding of the scale of contributed value from different ecosystem services, recording benefits in monetary terms also allows us to use other economic tools, such as economic valuation using welfare values, to measure the value of a specific project. An economic welfare analysis allows for justification of the overall investment (or similar investments in future, based on this project), and also facilitates the ability for different parties to co-fund investments based on the value of the ecosystem service changes that such a project produces.

Depending on the objective of a decision process, it is appropriate to use different definitions of monetary value. Throughout this Guide we provide a summary of different economic and accounting tools and valuation methods:

- Earlier, in Section 2, we introduce the concepts of exchange value for EEA (Section 2.1), welfare value for economic valuation (Section 2.2), other economic indicators (Section 2.3) and how to select which measure is appropriate for different types of decisions (Section 2.4).
- Figure 6.12 below provides an overview of the relevant methods available to estimate both exchange and welfare values for the different ecosystem services relevant to coastal wetlands and blue carbon restoration projects.
- Approaches to estimate exchange values for different ecosystem services are captured throughout the subsections that follow.

[88] Potschin -Young, M., Haines-Young, R., Görg, C., Heink, U., Jax, K., Schleyer, C. (2017). Understanding the role of conceptual frameworks: Reading the ecosystem service cascade. Ecosystem Services, 29:428-440. https://doi.org/10.1016/j.ecoser.2017.05.015

[89] See United Nations, et al. (2014). SEEA Central Framework, Section 3.2. https://seea.un.org/content/seea-central-framework

	Exchange values		Welfare values	
	Preferred methods	Proxy methods	Preferred methods	Proxy methods
First nations values			• Revealed preference • Stated preference	• Benefit transfer
Carbon sequestration	• Australian Carbon Credit Units • Reverse auction	• Certificate prices of Clean Development Mechanism • Averting behaviour • Avoided damage cost	• Social cost of carbon	• Stated preference
Water purification	• Expense for maintaining water quality • Expense for chemical removal	• Replacement cost • Avoided damage cost • Benefit transfer	• Revealed preference • Stated preference	• Benefit transfer
Coastal protection	• Defensive expenditure (e.g. construction cost of seawalls)	• Replacement cost • Avoided damage cost • Benefit transfer	• Revealed preference • Stated preference	• Avoided cost • Replacement cost • Benefit transfer
Fisheries biomass	• Market price		• Profit	
Recreational services	• Total trip expenditure	• Travel cost • Simulated exchange value • Benefit transfer	• Revealed preference • Stated preference	• Benefit transfer
Existence value			• Stated preference	• Benefit transfer

Figure 6.12: Valuation approaches for different ecosystem services.

Application at the site level

In order to estimate monetary values for a restoration project, at a minimum there is a need for an estimate of the level of activity or biophysical changes associated with the site, for example, visitation rates, or commercial production levels. This may require primary data collection. To then identify the monetary values associated with each 'unit' of activity (e.g., for each tonne of commercial fish extracted, or for each recreational trip made to a site) or other unit changes in ecosystem service outputs, requires further data. This would typically be acquired through primary data collection using surveys, and often these surveys are quite detailed and hence expensive to implement. For example, they may include surveys, of firm/individual level outputs/costs, or of representative samples of recreational users or the general public when estimating values associated with cultural services.

Such primary surveys to identify activity levels associated with ecosystem restoration may be technically feasible for sites with a sufficient stakeholder base to sample from, but could be relatively expensive to implement for small sites. Where primary data is not feasible to collect, use of average activity levels and average values for similar types of ecosystem services in similar locations could offer a more accessible alternative for compiling economic information. A more detailed discussion of extrapolating values through 'benefit transfer' is provided in Section 12.2. In some cases, where primary or secondary data is not available to estimate the monetary benefits of ecosystem services directly, we can instead refer to the costs avoided, for example through damage mitigation (also discussed in Section 12.2).

Identification of the monetary values enables ecosystem service accounts to be prepared, and total economic valuation or integrated economic assessment to be undertaken. An extension is to consider other economic indicators by using regional input output models to estimate regional multipliers for site specific expenditures; that is, direct expenditure in the region leads to additional expenditure by the recipients of that initial expenditure, leading to the multiplier effect. Regional multipliers are available in the literature and could be applied to give minimum (direct) and maximum (via multiplier) effects on the regional economy. For example, producing 1kg of prawns leads to approximately 5 times their initial value to the broader economy[90]. Multipliers are discussed further in Section 12.2.

[90] Voyer, M., et al. (2016). Social and Economic Evaluation of NSW Coastal Professional Wild-Catch Fisheries: Valuing Coastal Fisheries, University of Technology, Sydney, pg. 208. Report to Australian Fisheries Research and Development Corporation on Project 2014/301.

A key aspect of measures of economic benefits (as defined here) is one of additionality. One has to identify the increase in activity associated with the restoration, compared to the base. In cases where there is substitution of economic activity from alternative sites as result of the restoration, one cannot claim the full level of economic activity (production/jobs etc.) as a direct benefit associated with the change in value of the service flows at the restored site as there will have been reductions in activity from elsewhere. For example, aggregate combined expenditure by recreational fishers may not change as they shift site choice as a result of restoration: indeed, they may be reduced.

The issue of attribution is important when revealed preference approaches such as hedonic price models (discussed further in Section 12.2), which involve breaking down the characteristics of a good and attributing value to each characteristic, are employed, as otherwise there is a risk of double counting benefits that have been evaluated elsewhere.

Ecosystem services explored in this Guide

The following sections provide detailed discussion of the main ecosystem services that may change due to a restoration of a coastal blue carbon ecosystem:

- Carbon sequestration and emissions
- Water purification services
- Coastal protection: erosion, storm mitigation and flood control services.
- Fish production: nursery habitat services
- Fish production: biomass provisioning services
- Cultural services: recreation and non-use values
- Cultural services: First Nations values

7. Carbon stocks, sequestration & emissions

Dr Jeff Kelleway, Prof. Kerrylee Rogers, Emma Asbridge

7.1 Summary of section

Coastal wetlands are recognised for the disproportionate role they play in global carbon cycling, relative to their spatial extent, and are termed 'blue carbon' habitats[110]. This capacity of coastal wetlands to remove carbon from the atmosphere is based upon three main factors:

1. Coastal wetlands are productive ecosystems, meaning that plants and other primary producers draw large amounts of carbon dioxide from the atmosphere, which is stored aboveground and belowground in plant roots and rhizomes and in soils.

2. Frequent inundation by tidal waters reduces exposure of soils to oxygen, slowing the decomposition of organic matter, and leading to long-term storage of carbon. Meanwhile, saline waters minimise the production of methane, which is a powerful greenhouse gas and significant component of emissions from some freshwater settings.

3. There are substantial opportunities for creating and/or restoring coastal wetlands. Such actions can have the combined benefit of (1) reducing existing greenhouse gas emissions from degraded coastal landscapes; and (2) removing additional carbon dioxide from the atmosphere by the newly restored/created habitats where carbon is stored in plant biomass and soils (Figure 7.1).

Restoration activities in the coastal zone can influence the coastal carbon cycle in multiple ways. For example, interventions that modify the frequency, duration and/or seasonality of inundation may influence pre-existing (baseline) emissions of carbon dioxide, methane, and nitrous oxide. Changes to inundation regimes may also modify vegetation composition and productivity, rates of carbon decomposition, and the sedimentation or erosion or carbon-rich materials. Changes to water or soil chemistry may similarly influence vegetation composition and productivity, and/or the rates at which organic matter decomposes. Such changes may result in either positive or negative outcomes for greenhouse gas emissions, carbon sequestration and carbon stocks, with the direction and magnitude of outcomes dependent upon the land use transitions involved, extent, location, and timeframe of restoration actions.

[110] Macreadie, Peter I. et al. (2021). Blue Carbon as a Natural Climate Solution. Nature Reviews Earth & Environment, 2:826-839. https://doi.org/10.1038/s43017-021-00224-1

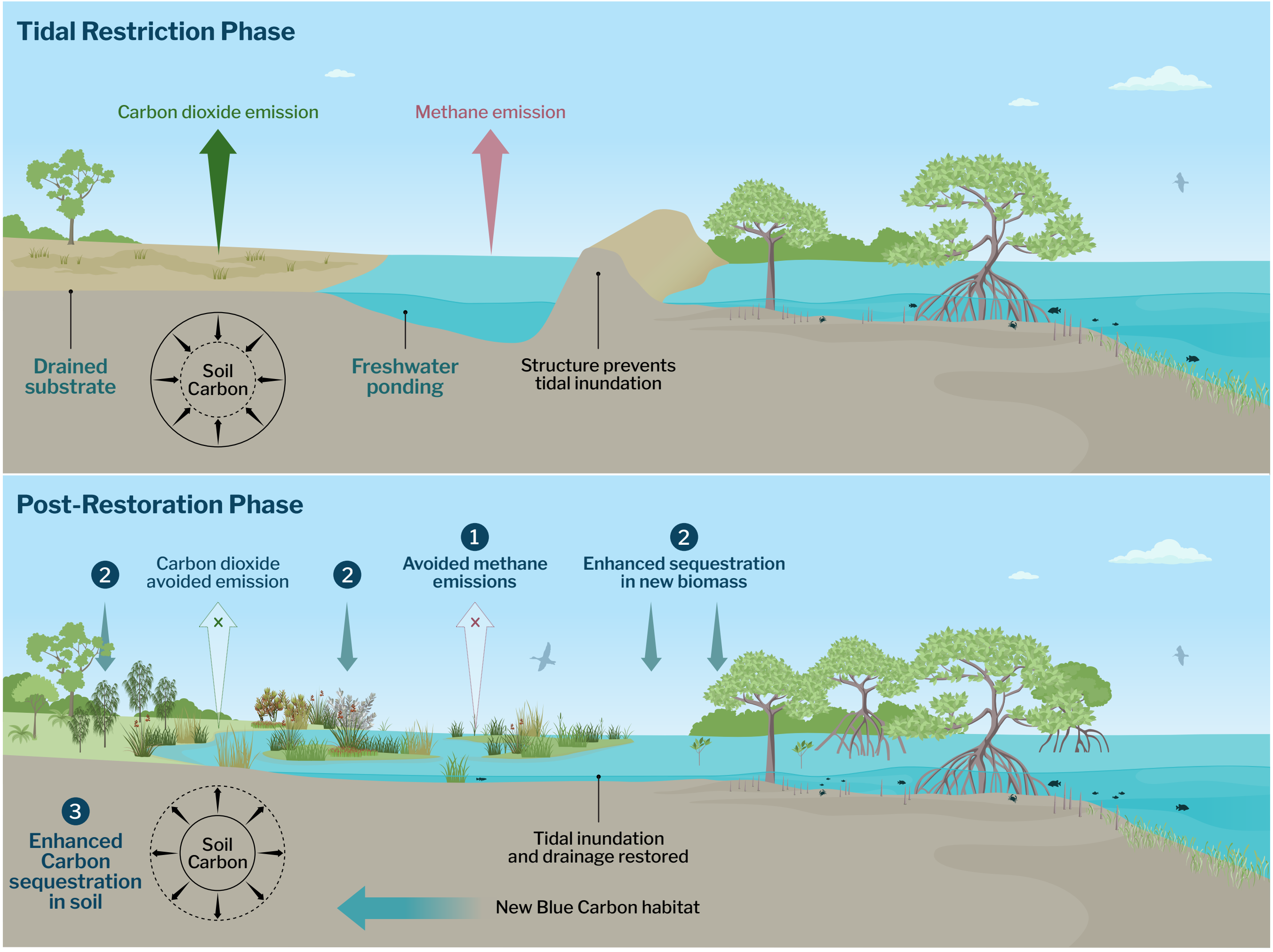

Figure 7.1: Carbon services conceptual figure.

Methods

This guide details an integrated approach for quantifying three related, though distinct, accounts associated with global climate regulation: (1) carbon abatement; (2) carbon sequestration; and (3) carbon stocks/storage. The carbon abatement account integrates estimates of greenhouse gas emissions and carbon sequestration through the life of a restoration project (i.e. years to decades) to determine the net outcomes of carbon abatement of tidal restoration actions at this site. This account includes both physical and financial accounts.

In contrast, the carbon stock/storage account provides snapshots of the amount of carbon stored in aboveground biomass and soil carbon (to 1 m depth) pools within the study area, estimated at two time points: (1) a pre-restoration time point; and (2) a post-restoration time point. No financial account has been estimated for carbon stocks as this would represent double-counting of values which are already considered in the carbon abatement account.

Project proponents can choose from two-tiers for estimating the physical accounts of carbon abatement, carbon sequestration, and carbon stocks/storage services, depending on their data and resource availability:

- **Nationally-consistent approach** – low-cost approach utilising existing nationally-available datasets and a variation of the Commonwealth Government's Blue Carbon Accounting Model (BlueCAM) calculator; or
- **Detailed approach** – integration of setting-specific and high-resolution datasets with the BlueCAM calculator to provide more accurate estimates of carbon accounts.

This guide provides methods and templates for the compilation of carbon abatement, carbon sequestration and carbon stock accounts over the life of a restoration project following the BlueCAM accounting framework[111]. Additional guidance is provided under the SEEA framework for the compilation of carbon storage and carbon sequestration accounts only (i.e. excluding emissions and net carbon abatement), which requires a separate estimation of single year accounts at the beginning and end of a project accounting period.

Data sources

The Blue Carbon Accounting Model (BlueCAM)[112] calculator is a repository of nationally relevant datasets and accounting procedures and is therefore a central tool for generating the carbon accounts described in this guide. BlueCAM requires some project specific parameters such as tidal range, elevation and land type/ecosystem extent accounts which are detailed in other sections of this guide. Where available, site-specific datasets of carbon cycling parameters may also be sourced from direct measurement, the literature, online repositories, or reliable unpublished sources to complement BlueCAM default values and provide more accurate estimates of carbon accounts.

7.2 Detailed section

Overall description of the topic

Coastal wetlands are recognised for the disproportionate role they play in global carbon cycling, relative to their spatial extent. While research over several decades has demonstrated the high productivity and carbon storage capacity of coastal ecosystems including seagrass, mangrove, tidal marshes, there has been increasing scientific and policy interest in the potential of these 'blue carbon' habitats to act as nature-based solutions to climate change over the past 10-15 years[113]. The capacity of coastal wetlands to remove carbon from the atmosphere is based on three main factors:

1. Coastal wetlands are typically productive ecosystems, whereby plants and other primary producers draw large amounts of carbon dioxide from the atmosphere, which is stored aboveground in plant biomass and belowground in plant roots and rhizomes and in soils.

2. Frequent inundation by tidal waters reduces exposure of soils to oxygen, slowing the decomposition of organic matter, and leading to long-term storage of carbon. Meanwhile, saline waters typically minimise the production of methane, which is a powerful greenhouse gas and significant component of emissions from some freshwater settings.

3. There are substantial opportunities for creating and/or restoring coastal wetlands. Such actions can have the combined benefit of (1) reducing existing greenhouse gas emissions from degraded coastal landscapes; and (2) removing additional carbon dioxide from the atmosphere by the newly restored/created habitats where carbon is stored in plant biomass and soils.

Broadly, the potential for blue carbon climate mitigation benefits can be achieved through minimising/stopping activities which release stored blue carbon (e.g. habitat loss, dredging, deforestation), or through the restoration and/or creation of new blue carbon habitat in areas that

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[111] This Guide is not directly applicable to the preparation of projects under the ACCU Scheme's Tidal Restoration of Blue Carbon Ecosystems method, nor for the management and reporting requirements of approved projects under that method. Appropriate guidance is provided at Tidal restoration method

[112] See Clean Energy Regulator (2022). The blue carbon accounting model (BlueCAM). Clean Energy Regulator. https://www.cleanenergyregulator.gov.au/DocumentAssets/Pages/The-blue-carbon-accounting-model-BlueCAM.aspx; AND Clean Energy Regulator (2022). Tidal restoration of blue carbon ecosystems method. Clean Energy Regulator. https://www.cleanenergyregulator.gov.au/ERF/Choosing-a-project-type/Opportunities-for-the-land-sector/Vegetation-methods/tidal-restoration-of-blue-carbon-ecosystems-method

[113] Macreadie, Peter I. et al. (2021). Blue Carbon as a Natural Climate Solution. Nature Reviews Earth & Environment, 2:826-839. https://doi.org/10.1038/s43017-021-00224-1

have historically been degraded[114]. Restoration activities can alter carbon services via changes to carbon stocks, carbon sequestration, and their emissions of carbon dioxide and other greenhouse gases, including methane and nitrous oxide (Table 7.1). Carbon can be stored in many components of an ecosystem, termed pools, and the largest pools in coastal wetlands include (1) roots and soil carbon stored belowground, and (2) living aboveground biomass. Carbon sequestration refers to the rate at which carbon is added to these pools.

This section provides methods and templates for the compilation of inter-related accounts of (1) carbon abatement, (2) carbon sequestration and (3) carbon stock accounts over the life of a restoration project following the BlueCAM accounting framework. Additionally, guidance is provided for the compilation of carbon storage and carbon sequestration accounts only (i.e. excluding avoided emissions and net carbon abatement) as per the SEEA framework. The SEEA framework also differs from BlueCAM in that it requires separate estimation of single year accounts at the beginning and end of a project accounting period. Methods are detailed for the computation of these single year accounts to enable populating of SEEA carbon storage and carbon sequestration accounts.

Table 7.1: Terminology and definitions relating to carbon services and their inclusion within existing accounting frameworks

Terminology	Description	Incorporated in BlueCAM?	Incorporated in SEEA framework?
Carbon stock	The quantity of carbon stored within specified 'pools' at one point in time. This stock may reflect carbon accumulated over timescales extending to decades or centuries for living biomass, and hundreds to thousands of years of accumulation in soil carbon pools. A change in stock will result as the net balance of new sequestration and any actual (but not avoided) emissions.	Not a direct output, but can be determined through steps outlined below	Yes, though referred to as '*carbon storage*'
Carbon sequestration	The rate at which new carbon is removed from the atmosphere and added to long-term storage pools (e.g. 'biomass' and 'soil' carbon pools).	Yes	Yes
Emissions/ avoided emissions	Emissions are the greenhouse gases emitted from soils, water and vegetation. Emissions of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are typically accounted for, but reported in carbon dioxide equivalents (CO_2e) Avoided emissions refer to the net decrease in greenhouse gas emissions which are likely to have occurred if the restoration actions had not been undertaken.	Yes	Actual emissions = incorporated as carbon storage reduction. Avoided emissions = No
Carbon ebatement	Carbon abatement is the net balance of avoided emissions and carbon sequestration (if any) occurring under baseline (pre-restoration) land use conditions and sequestration and emission of greenhouse gases in a specified reporting period following restoration activities	Yes	No

[114] Kelleway, J. J., et al. (2020). A National Approach to Greenhouse Gas Abatement through Blue Carbon Management. Global Environmental Change, 63:102083. https://doi.org/10.1016/j.gloenvcha.2020.102083

Restoration activities in the coastal zone can influence the coastal carbon cycle in multiple ways. For example, interventions that modify the frequency, duration and/or seasonality of inundation may influence pre-existing (baseline) emissions of carbon dioxide, methane and nitrous oxide. Changes to inundation regimes may also modify vegetation composition and productivity, rates of carbon decomposition, and the sedimentation or erosion or carbon-rich materials. Changes to water or soil chemistry may similarly influence vegetation composition and productivity, and/or the rates at which organic matter decomposes. Such changes may result in either positive or negative outcomes for greenhouse gas emissions, carbon sequestration and carbon stocks, with the direction and magnitude of outcomes dependent upon the land use transitions involved, extent, location and timeframe of restoration actions.

Several accounting frameworks are available to estimate climate mitigation capacity of coastal landscapes and ecosystems. These range in scale from national and international accounting frameworks (e.g. IPCC Wetland Supplement[115]), through to project-scale accounting for carbon credits on voluntary trading[116] or government-regulated[112] schemes. Frameworks may vary in their terminology and coverage of carbon and greenhouse gas fluxes, as demonstrated for the SEEA climate regulation accounting approach[117] and Blue Carbon Accounting Model (BlueCAM)[118] (Table 7.2). The BlueCAM framework represents the most comprehensive and suitable approach for accounting for Blue Carbon in Australian restoration projects and is therefore the central focus of accounting methods described in this guide, including as a basis for the estimation of SEEA climate regulation accounts.

Methods and data sources

This guide details an integrated approach for quantifying three related, though distinct, accounts associated with greenhouse gas regulation service provision: (1) *carbon abatement*; (2) *carbon sequestration*; and (3) *carbon stocks/storage* (Table 7.2). The carbon abatement account integrates estimates of actual and avoided greenhouse gas emissions and carbon sequestration through the life of a restoration project to determine the net outcomes of carbon abatement of tidal restoration actions at this site. This account includes both physical and financial accounts.

In contrast, the *carbon stock/storage* account provides snapshots of the amount of carbon stored in aboveground biomass and soil carbon (to 1 m depth) pools within the study area, estimated at two discrete time points: (1) a pre-restoration time point; and (2) a post-restoration time point (typically as recent as datasets allow). No financial account has been estimated for carbon stocks as this would represent a double-counting of values which are already considered in the carbon abatement account derived from BlueCAM (and the carbon sequestration account derived for SEEA account purposes).

Project proponents can choose from two tiers for estimating the physical accounts of *carbon abatement*, *carbon sequestration*, and *carbon stocks/storage services*, depending on their data and resource availability (Table 7.2):

- **Nationally-consistent approach** low-cost approach utilising existing nationally-available datasets and a variation of the Commonwealth Government's Blue Carbon Accounting Model (BlueCAM) calculator; or

[115] Hiraishi, T., et al. (Eds.). (2014). 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Switzerland: IPCC.

[116] Needelman, B. A., Emmer, I. M., Emmett-Mattox, S., Murray, L. A., Ruggiero, P., Crooks, S., & Sanderman, J. (2018). The Science and Policy of the Verified Carbon Standard Methodology for Tidal Wetland and Seagrass Restoration. Estuaries and Coasts, 41:2159–2171. https://doi.org/10.1007/s12237-018-0429-0

[117] Keith, H., et al. (2021). Evaluating nature-based solutions for climate mitigation and conservation requires comprehensive carbon accounting. Science of The Total Environment, 769:144341. https://doi.org/10.1016/j.scitotenv.2020.144341

[118] Clean Energy Regulator (2022). Tidal restoration of blue carbon ecosystems method. Clean Energy Regulator. https://www.cleanenergyregulator.gov.au/ERF/Choosing-a-project-type/Opportunities-for-the-land-sector/Vegetation-methods/tidal-restoration-of-blue-carbon-ecosystems-method

- **Detailed approach** – integration of setting-specific and high-resolution datasets requiring primary data collection with the BlueCAM calculator to provide more accurate estimates of carbon accounts.

This guide provides methods and templates for the compilation of carbon abatement, carbon sequestration and carbon stock accounts over the life of a restoration project following the BlueCAM accounting framework. Additionally, guidance is provided for the compilation of carbon storage and carbon sequestration accounts only (i.e. excluding emissions and net carbon abatement) under the SEEA framework, which requires separate estimation of single year accounts at the beginning and end of a project accounting period.

A full workflow integrating BlueCAM and SEEA frameworks – with differentiation between nationally-consistent and detailed approaches – is summarised in Figure 7.2.

Both approaches require inputs from the Ecosystem Extent accounts detailed above, a minor element of spatial analysis (additional to that of the extent accounts), and generation of multiple model runs of the BlueCAM calculator. The time commitment required to undertake a desktop-based, nationally-consistent approach may extend to multiple days, depending on the complexity of the restoration project and familiarity of the operator with BlueCAM. Compilation of settings-specific datasets and/or the collection of new site-specific datasets will require additional time and resources, which should be considered on a case-by-case basis. Familiarity with BlueCAM and experience working with blue carbon datasets would be particularly beneficial when undertaking the detailed-approach.

Table 7.2: Summary of two tiers of approach described for carbon abatement and carbon stock accounts.

Approach	Datasets used	Expected outcome
Nationally-consistent approach	- Nationally-consistent extent accounts - Nationally-consistent tide gauge approach - Nationally-available elevation dataset - Modified version of BlueCAM calculator with additional outputs for EEA projects	Low site-specificity ⟶ less reliable account
Detailed approach	- Detailed approach extent accounts - Site-specific tide gauge approach - Nationally-available elevation dataset - Site-specific and/or setting-specific (same or similar estuaries) blue carbon datasets, from published and unpublished sources - Modified version of BlueCAM calculator with additional outputs for EEA projects	High site-specificity ⟶ highly reliable account

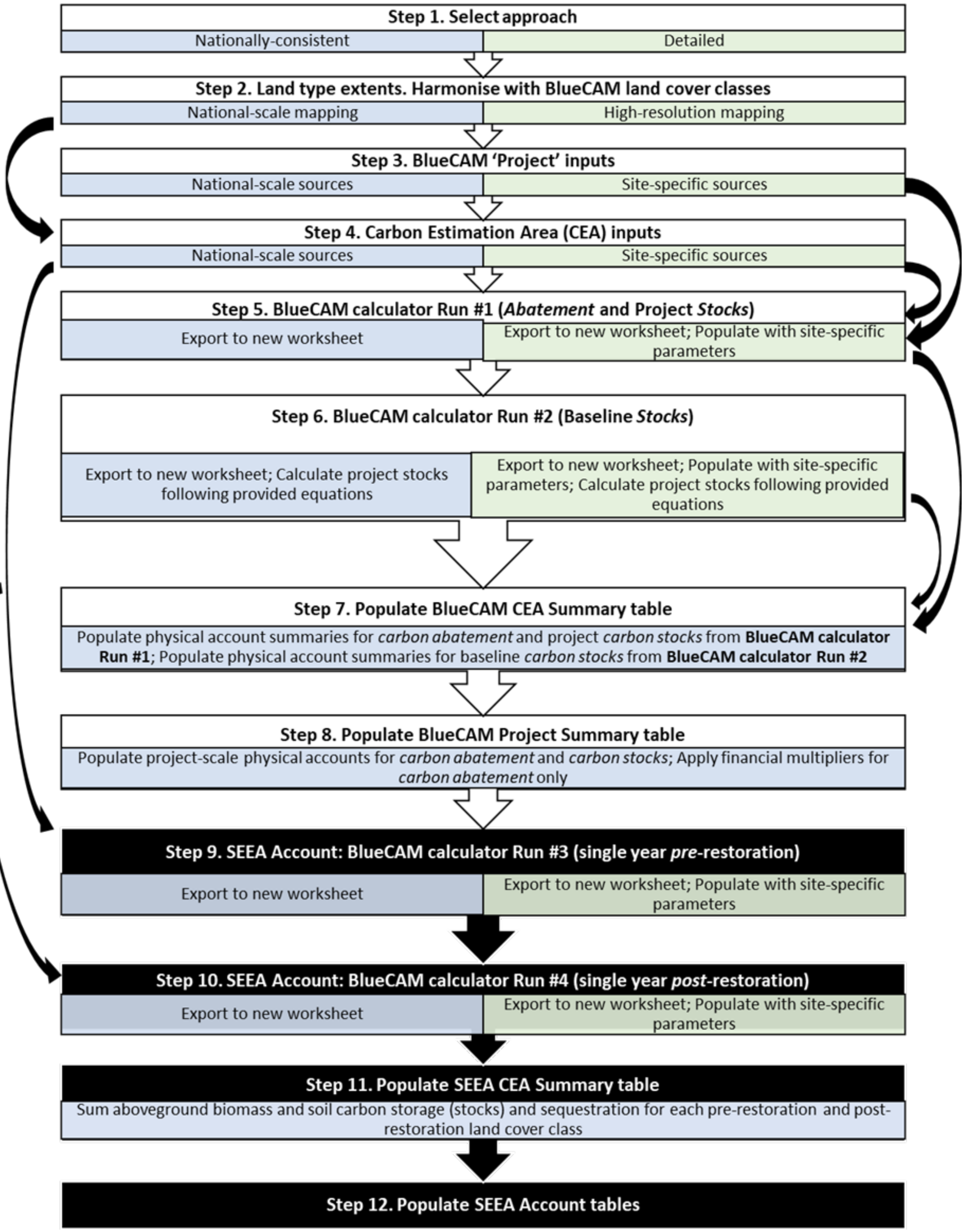


Figure 7.2: **Flow chart of steps required to quantify carbon abatement and carbon stock accounts using a BlueCAM approach (steps 1-8), and extension to SEEA accounts for carbon storage and carbon sequestration (steps 9-12).** Green shading represents steps where detailed approach varies from the default nationally-consistent approach. Red text refers to instructional or data-template tables contained in this section.

BlueCAM framework

The Blue Carbon Accounting Model (BlueCAM)[118] is a tool developed by a scientific working group of national blue carbon experts and was released by the Commonwealth government in 2022. It was created for the specific purpose of quantifying carbon abatement, and awarding of carbon credits, for projects registered under the Australian Carbon Credit Units (ACCU) Scheme (formally known as the Emission Reduction Fund)

The current operational version of BlueCAM is the 'BlueCAM calculator' - a Microsoft Excel workbook (password protected), which acts as a repository of nationally-relevant datasets and accounting procedures. For this reason, the BlueCAM calculator is the most relevant and useful tool available to serve as a foundation for the accounting of carbon services under the current Environmental Economic Accounts framework. Extensive technical documentation on the datasets, models and assumptions of BlueCAM are available here:

- ACCU Scheme methods
- Blue Carbon Accounting Model (BlueCAM) Guidelines
- Blue Carbon Accounting Model (BlueCAM) Technical Overview

Further detail is also available in the following scientific publications authored by the expert working group:

- Lovelock, C.E., Adame, M.F., Bradley, J., Dittmann, S., Hagger, V., Hickey, S.M., Hutley, L.B., Jones, A., Kelleway, J.J., Lavery, P.S., Macreadie, P.I., Maher, D.T., McGinley, S., McGlashan, A., Perry, S., Mosley, L., Rogers, K. & Sippo, J.Z. (2022). An Australian blue carbon method to estimate climate change mitigation benefits of coastal wetland restoration. Restoration Ecology e13739. https://doi.org/10.1111/rec.13739
- Lovelock, C. E., Adame, M. F., Butler, D. W., Kelleway, J. J., Dittmann, S., Fest, B., King, K. J., Macreadie, P. I., Mitchell, K., Newnham, M., Ola, A., Owers, C. J. & Welti, N. (2022). Modelled approaches to estimating blue carbon accumulation with mangrove restoration to support a blue carbon accounting method for Australia. Limnology and Oceanography, 67:S50-S60. https://doi.org/10.1002/lno.12014

The methodology for all tiers of approach (i.e. nationally-consistent approach, detailed approach, and variations explored in the Hunter and East Trinity case studies) uses the BlueCAM calculator, and these approaches broadly follow the requirements of the following BlueCAM guidance documents and scientific outputs referred to above.

In some instances, variations from these guidance documents are implemented in this Guide, for the following reasons: (1) to provide greater simplicity for higher level EEA assessments (i.e. as opposed to ERF projects which seek ACCUs); (2) to ensure consistency with other physical and financial accounts quantified in the case study; and (3) to enable use of setting-specific datasets in the detailed approach. The rationale for such variations from BlueCAM guidance is provided in this section and tables below.

Note: The publicly-available BlueCAM calculator file can be used to generate all required outputs for the carbon abatement account (referred to throughout this Guide as 'Run #1'). This public version of the calculator, however, does not provide the outputs required for the carbon stock account – in this instance an additional, modified version of BlueCAM for EEA purposes is required (referred to throughout this Guide as 'Run #2'). In both instances the active worksheets of the publicly-available BlueCAM calculator file need to be copied and pasted to a new blank Excel document to allow access to columns hidden from view in the official ERF file version. This step is particularly important if following the 'detailed approach' whereby default values of the ERF file version can be replaced by setting-specific values, if appropriate.

It is also important to note that due to the modifications referred to above, the accounting framework detailed in this Guide for EEA purposes is not adequate for the purposes of carbon-credit accounting under the ACCU Scheme. Such projects should follow the explicit methods, instructions and requirements detailed in the links above.

BlueCAM inputs (Steps 2 - 4):

Operation of BlueCAM for both carbon abatement and carbon stock accounting purposes requires two types of data inputs: (1) project level parameters; and (2) Carbon Estimation Area (CEA) parameters.

Project level parameters include project accounting timeframes, the tidal range of the project site, and quantification of any fuel use associated with the project. The sources for these project level parameter inputs, and rationale for their use for is detailed in Table 7.3.

Accurate carbon abatement accounting may require the stratification of the project area into sub-units (termed Carbon Estimation Areas or CEAs in BlueCAM). For BlueCAM, CEAs may need to be delineated within a project area based on different land-uses, vegetation types and levels of land elevation (relative to Australian Height Datum or m AHD) – factors which may all change for a given parcel of land over the life of a project. While ERF projects are typically required to monitor and delineate CEAs at multiple intervals (e.g. every five years) over the life of a project, carbon accounting for EEA purposes uses a simplified approach. That is, CEAs are delineated based on just two time points: a CEA baseline land type based upon status prior to restoration actions being applied (baseline); and a reporting period (post-restoration) status. The exact dates of baseline and reporting timeframes will vary among projects and may be influenced the timing and availability of key datasets (e.g. extent mapping). The source of CEA parameter inputs, and rationale for their use is detailed in Table 7.4. Further guidance on the definition of CEA land types is provided in ERF technical documents.

Spatial analyses are required to determine the number, type and extent of each CEA. Relevant ecosystem extent mapping layers for CEA baseline land type and post-restoration land type can be derived from the Ecosystem Extent accounts workflow (Step 2). These are then used as inputs in a 'change detection analysis' in a GIS platform[119]. Change detection analysis returns a new raster layer depicting the extent of each category of land type change within the project area. If the land types defined by the extent account approaches do not align perfectly with the prescribed land type inputs available in BlueCAM, a harmonisation process is

[119] See for example: https://pro.arcgis.com/en/pro-app/latest/help/analysis/image-analyst/change-detection-in-arcgis-pro.htm

required whereby input land classes need to be converted to the most suitable BlueCAM land type class (see Hunter and East Trinity case studies for examples of the harmonisation process).

The elevation of a CEA operates as a modifier of some carbon cycling parameters in BlueCAM. To determine the elevation of each CEA, a further spatial analysis is required. In this instance, the change detection output raster above is converted to multipart polygon files[120], and a zonal statistics tool[121] is used to compute central estimates of elevation for each polygon/CEA, using a high-resolution digital elevation model. In most instances, the median elevation value for each CEA will be the most appropriate selection as this is less sensitive than the mean value to outliers at either upper or lower elevation ranges.

BlueCAM model runs #1 and #2 (Steps 5-6): Setting-specific data sources (higher cost detailed approach)

Finalised project level parameters and CEA input parameters are entered into the relevant climatic zone worksheet of a new publicly-available BlueCAM calculator file following specifications outlined for nationally-consistent or detailed approaches in Table 7.3 and Table 7.4.

Where available, site-specific datasets of carbon cycling parameters (Table 7.5) may be sourced from direct measurement, scientific literature, online repositories, or reliable unpublished sources to complement BlueCAM default values and provide more accurate estimates of carbon accounts. Such datasets should only be used in systems that are in the same or similar region as the project area, share similar geomorphic, hydrologic, and biological properties, and are under similar management regimes, unless any differences should not have a substantial effect on GHG emissions[122].

Where resourcing allows, project-specific carbon assessments can also be improved by collecting and applying site-specific data derived from field measurements of aboveground biomass, belowground carbon stocks, and/or measurement of greenhouse gas fluxes. This can be achieved using numerous approaches[123] including the following for stock and sequestration parameters:

- A stock change approach includes repeated measurements of biomass in the vegetation or soils carbon pool across multiple time periods. In this case, this would involve assessing soil and vegetation carbon stocks before and after restoration.
- A space-for-time approach presumes that carbon stocks within pools are proportional to their age. Quantifying the age of various carbon pools allows for stock change over time to be determined. The age of carbon pools can be estimated from extent change accounts, as detailed in that section.
- Carbon stocks can be estimated using standardised approaches to measure carbon stocks in the vegetation and soil carbon pools for each coastal ecosystem vegetation community. Where high resolution data is available on variation in the structure of vegetation (e.g. canopy height models), modelled approaches may be used to quantify aboveground biomass and improve the accuracy of carbon stock accounts.

Direct measurement of greenhouse gas fluxes (including carbon dioxide, methane and nitrous oxides) may greatly reduce uncertainties associated with BlueCAM-derived emissions and avoided emissions estimates. Measurement protocols should follow best-practice procedures contained in the scientific literature, but include 1) Chambers 2) Flux towers 3) Dissolved concentrations (head space method)[124,125],. While this direct measurement is likely to require

[120] See for example: https://pro.arcgis.com/en/pro-app/latest/tool-reference/conversion/raster-to-polygon.htm

[121] See for example: https://pro.arcgis.com/en/pro-app/latest/tool-reference/spatial-analyst/zonal-statistics-as-table.htm

[122] Restore America's Estuaries & Silvestrum (2015). VM0033 Methodology for Tidal Wetland and Seagrass Restoration. Sectoral Scope 14, Verified Carbon Standard.

[123] Howard, J., Hoyt, S., Isensee, K., Pidgeon, E., Telszewski, M. (eds.) (2014). Coastal Blue Carbon: Methods for assessing carbon stocks and emissions factors in mangroves, tidal salt marshes, and seagrass meadows.

specialised equipment and personnel, advances are being made in low-cost sensors and satellite technology which might be utilised to improve cost-effectiveness[126].

In all instances, parameters must be converted to the appropriate units, and may require specific treatment (Table 7.5) prior to insertion in the relevant CEA line item of the modified BlueCAM calculator file. Any other data conversions (e.g. conversion of biomass and/or soil organic matter units to carbon estimates; conversion of non-CO_2 fluxes to CO_2e values) should be consistent with methods detailed in BlueCAM technical documents and associated publications.

Calculation of carbon stocks (Step 5-6)

While the publicly available BlueCAM calculator file incorporates default values for carbon stocks in biomass and soil carbon pools, it does not present these as outputs for the timeframes relevant to 'opening' and 'closing' (i.e. baseline and post-restoration time points) accounts of EEA frameworks. Consequently, these output values need to be created manually via the use of two modified BlueCAM calculator files (i.e. Runs #1 AND #2) as detailed in Table 7.6 and Table 7.7. This approach estimates the stock of both aboveground biomass and soil carbon pools under both baseline conditions (via Run #2) and post-restoration conditions (sum of baseline stock estimates from Run #2 plus specified parameters from Run #1).

BlueCAM outputs (Steps 7-8): Physical accounts

The following steps apply equally to both nationally-consistent and detailed approaches:

Carbon abatement: Two sets of outputs are derived from BlueCAM calculator to populate carbon abatement account tables. These are: (1) estimates of carbon abatement parameters for each individual CEA (template: Table 7.8), populated from BlueCAM calculator Run #1 rows AC, AG and AM; and (2) project level abatement estimates (template: Table 7.9) populated from BlueCAM calculator Run #1 cells AQ3: AT3. Note that BlueCAM automatically applies at 5% reduction on the overall abatement estimate (i.e. Net abatement amount (Ar)) within the BlueCAM calculator (i.e., cell AT3). This discount is a specific requirement of projects seeking carbon credits under the tidal restoration method of the ACCU Scheme but is less relevant to EEA projects which are not operating under the ACCU Scheme. For this reason, Table 7.8 includes an additional row 'Net abatement amount (Ar-adj): ERF discount removed', whereby Ar-adj is the net sum of values EA, CP and Efk (i.e., no 5% discount applied).

Carbon stocks: Four carbon stock parameters are derived across Run #1 and Run #2, as specified by Table 7.7, with reporting templates provided at the scale of individual CEAs (Table 7.7) and project scale stocks (Table 7.9).

Financial accounts (Carbon abatement only):

BlueCAM net abatement estimation outputs (i.e. columns AQ:AT in BlueCAM worksheet, plus the Ar-adj value described above) are used to populate the overall physical account estimates for carbon abatement in Table 7.8. Each of these high-level physical account estimates – in Tonnes CO_2e and calculated over the life of the restoration accounting period (i.e. pre-restoration baseline year to post-restoration reporting year) were also used to calculate related financial accounts. Two financial account approaches are demonstrated. The first approach applies an Australian Carbon Credit Unit spot price value of $30.75 per Tonne of CO_2e abatement, as reported by the Clean Energy Regulator in the Quarterly Carbon Market Report – September Quarter 2022. A second approach using a financial multiplier of $150 per Tonne of CO_2e abatement can be applied to reflect the expectation that carbon credits generated in Blue Carbon projects are likely to attract a premium market value (relative to other carbon credits) due to their multiple co-benefits and high market

[124] Bussman et al. 2022. Spatial variability and hotspots of methane concentrations in a large temperate river. Front. Environ Sci. https://www.frontiersin.org/articles/10.3389/fenvs.2022.833936/full

[125] Clough et al. 2007. Diurnal fluctuations of dissolved nitrous oxide (N2O) concentrations and estimates of N2O emissions from a spring-fed river: implications for IPCC methodology. https://onlinelibrary.wiley.com/doi/full/10.1111/j.1365-2486.2007.01337.x

[126] Hondula, K. L., DeVries, B., Jones, C. N., & Palmer, M. A. (2021). Effects of using high resolution satellite-based inundation time series to estimate methane fluxes from forested wetlands. Geophysical Research Letters, 48(6), e2021GL092556.

Table 7.3: **Project-level BlueCAM input parameters, their descriptions and rationale for use in restoration project carbon abatement and carbon stock accounting.** Further guidance on each BlueCAM parameter is provided in ERF technical documentation.

<table>
<tr><th rowspan="2">Project information parameter</th><th colspan="2">Input description / rationale</th><th rowspan="2">Source / Links</th></tr>
<tr><th>Nationally-consistent approach</th><th>Detailed approach[1]</th></tr>
<tr><td>Climatic zone</td><td colspan="2">Climate: BlueCAM uses climatic regions aligned with the Australian Government's Natural Resource Management (NRM) regions approach for projecting the influence of climate change to estimate regionally specific abatement.</td><td>Link: NRM regions</td></tr>
<tr><td>Reporting period start date (day/ month/year)</td><td colspan="2">Date of commencement of restoration activities</td><td>Project restoration timeline</td></tr>
<tr><td>Reporting period end date (day/ month/year)</td><td colspan="2">As defined in the guide, EEA projects can select a project accounting period of either 25 years or 100 years. Projects with a permanence period of 25 years (and projects with a 100 year permanence period which are subject to the project area discount) are subject to a 25% reduction in carbon abatement estimates, which is applied automatically by BlueCAM.

A project accounting period of 100 years, with no project area discount should be selected for EEA purposes</td><td></td></tr>
<tr><td>Apply project area discount?</td><td colspan="2">Input = 'No'

A project accounting period of 100 years, with no project area discount should be selected for EEA purposes.</td><td></td></tr>
<tr><td rowspan="2">Enter the tidal range (m)</td><td colspan="2">The distribution of coastal wetland types, their carbon cycling parameters, and responses to anticipated sea-level rise are influenced by tidal inundation parameters. Input value (m) = Highest astronimical tide (m) - Lowest astronomical tide (m)</td><td rowspan="2">Link: BoM[2]</td></tr>
<tr><td>Tidal range data for the nearest / most suitable public tidal gauge for the restoration area</td><td>Site-specific tidal range data for the restoration site from available sources such as: (1) water level measurements within the post-restoration setting; (2) post-restoration specifications of engineering structures which control inundation</td></tr>
<tr><td>Fuel consumed during reporting period</td><td>A general principle in many carbon accounting frameworks is that carbon pools or emissions which represent less than 5% of overall project abatement may be considered 'de minimis'. For the purpose of EEA reporting, fuel consumption may be assumed to be zero for project activities (where it is reasonable to assume these emissions represent less than 5% of overall project abatement).</td><td>Project-specific estimates of fuel consumption related to project activities, if available.</td><td>Project works records, plant and vehicle logs.</td></tr>
</table>

Table 7.3: cont.

Project information parameter	Input description / rationale		Source / Links
	Nationally-consistent approach	Detailed approach[1]	
Carry over net abatement from the previous reporting period	A single reporting period was used for EEA estimation purposes, therefore no value is entered here.		

[1] Where detailed information is unavailable then default to the National-approach input (acknowledging the limitations and uncertainty which may be associated with this)

[2] More locally-relevant tide gauge and water level datasets may be available from sub-national data repositories, including those listed below by state:

QLD https://www.msq.qld.gov.au/tides/open-data

NSW https://www.mhl.nsw.gov.au/Data-Level

VIC https://www.vicports.vic.gov.au/community-and-bay-users/Pages/Waves-wind-weather.aspx

SA https://water.data.sa.gov.au/

WA https://www.transport.wa.gov.au/imarine/tide-data-real-time.asp

NT https://ntg.aquaticinformatics.net/

Table 7.4: **Carbon Estimation Area (CEA) BlueCAM input parameters, their descriptions and rationale for use in restoration project carbon abatement accounting.**

CEA parameter	Input description / rationale		Source / Links
	Nationally-consistent approach	Detailed approach[1]	
CEA area (ha)	Area of each unique change (pre-restoration to post-restoration) class, as determined from national extent mapping approach	Area of each unique change (pre-restoration to post-restoration) class, as determined from detailed extent mapping approach	Ecosystem extent maps
Elevation of CEA (m AHD)	Median elevation value of all pixels within CEA, as derived from nationally-available, high resolution DEM	Median elevation value of all pixels within CEA, as derived from best-available, high-resolution DEM	Detailed DEM
Tidal introduction in CEA?	**Input = 'Yes'** for CEAs within the limits of the restoration extent mapping and with CEA median elevation estimates within the range of LAT to HAT values from the best available tide gauge **Input = 'No'** for CEAs which have CEA median elevation estimates outside the range of LAT to HAT values from the best available tide gauge	**Input = 'Yes'** for CEAs within the limits of the restoration extent mapping and with CEA median elevation estimates within the range of LAT to HAT values specific to the restoration project **Input = 'No'** for CEAs which have CEA median elevation estimates outside the range of LAT to HAT values specific to the restoration project	Ecosystem extent maps; Detailed DEM Tidal range and tidal plane (LAT, HAT) estimates; CEA median elevation estimates

Table 7.4: cont.

CEA parameter	Input description / rationale		Source / Links
	Nationally-consistent approach	Detailed approach[1]	
New CEA or first reporting period?	**Input = 'Yes'** for CEAs which experience a change in land type between baseline and end of accounting periods **Input = 'No'** for CEAs which experience no change in land type between baseline and end of accounting periods	**Input = 'Yes'** for CEAs which experience a change in land type between baseline and end of accounting periods **Input = 'No'** for CEAs which experience no change in land type between baseline and end of accounting periods	Ecosystem extent maps Ecosystem extent maps
CEA baseline land type	Derived from pre-restoration national extent account approach. Land type may need to be harmonised with BlueCAM-specific land types	Derived from pre-restoration detailed extent account approach. Land type may need to be harmonised with BlueCAM-specific land types	Ecosystem extent maps
Land type for CEA: last reporting period end	N/A (only one reporting preiod used)		
Land type for CEA: current reporting period end	Derived from pre-restoration national extent account approach. Land type may need to be harmonised with BlueCAM-specific land types	Derived from pre-restoration detailed extent account approach. Land type may need to be harmonised with BlueCAM-specific land types	Ecosystem extent maps
Age of blue carbon vegetation in previous reporting periods (years)	Age of vegetation at baseline. Use historical records including aerial photograph and satellite imagery where available. If this information is not available, then an input value of 20 years can be assumed to be reflective of mature vegetation carbon stocks		Historical records, aerial and satellite imagery
Age of blue carbon vegetation in current reporting period (years)	Age of vegetation at reporting period. Use historical records including aerial photograph and satellite imagery where available. If this information is not available then assume: **Input = reporting year - baseline year** for vegetation that has emerged following restoration intervention (i.e. assumes vegetation change commenced immediately following intervention) **Input = reporting year - baseline year + baseline vegetation age** for pre-restoration vegetation that has persisted following restoration intervention (i.e. areas of no change in land cover class)		Historical records, aerial and satellite imagery; vegetation age at baseline (above parameter)
Excavation area within CEA (hectares)	Estimate from site work plans and records. Assume zero if no excavation or very minimal excavation undertaken		Project work plans and records

[1] Where detailed information is unavailable then default to the National-approach input (acknowledging the limitations and uncertainty which may be associated with this)

demand.

SEEA accounts (Steps 9-12):

Physical accounts of carbon stocks (termed 'storage' under the SEEA framework) for 'opening' and 'closing' periods can be transferred from the pre-restoration and post-restoration BlueCAM outputs which are reported at the scale of individual CEAs in Table 7.7. To derive stock estimates for specific ecosystems, opening accounts should be summed according to relevant 'baseline land types' (e.g. all CEAs which are 'mangrove' in the baseline should be summed for the mangrove ecosystem opening account), while closing accounts should be summed according to the relevant 'current reporting period end' land types.

The BlueCAM approaches described in the sections above do not provide opening and closing sequestration accounts, which are required by the SEEA framework. An additional approach was therefore undertaken whereby two further BlueCAM model runs were undertaken: (1) a simulation of a single year prior to commencement of restoration (i.e. Run # 3) and (2) a simulation of a single year at the end of the post-restoration accounting period (i.e. Run # 4). Inputs for each of these additional simulations should follow CEA parameters (including modification of any setting-specific data inputs) relevant to baseline and end of project scenarios, respectively.

Key assumptions or limitations

BlueCAM was developed specifically for the quantification of carbon sequestration and greenhouse gas emissions associated with tidal restoration projects in coastal wetland under the ACCU Scheme. In this context, BlueCAM is applied to ecosystems including mangroves, tidal marshes, seagrasses, mudflats and supratidal forests and a variety of land cover types which may present in coastal areas subject to tidal restriction. As such, the quantification parameters are most relevant to restoration interventions where removal or modification of tidal barriers has been undertaken, though will be useful to most types of restoration projects which involve changes to the extent or character of these land cover types. In instances

where a restoration project involves land cover types not currently included within BlueCAM a decision may be made to utilise the most relevant land cover type available, or to utilise setting-specific datasets[127].

While not required, project developers may wish to directly measure the carbon sequestration occurring on site (or use existing datasets, where available) if they believe the carbon abatement may exceed that estimated by BlueCAM. While direct measurement will not generate carbon credits (as present methods are restricted to quantifications via BlueCAM for registered projects), undertaking direct measurement may be beneficial to filling knowledge gaps (including improvements for future accounting approaches), and may be relevant for restoration activities not covered by the ERF tidal restoration method.

The detailed approach provides an opportunity for development of new site-specific datasets and insights by a project proponent or contractor. Generation of this data will require relevant field equipment, laboratory facilities and expertise. Uptake of this approach is therefore likely to be limited to well-resourced projects with such expertise, or those operating in collaboration with practitioners/researcher that have experience in the collection and analysis of such datasets. While the detailed approach is resource intensive, it provides a framework for the collection of high-quality and consistent new data that will likely improve understanding of restoration outcomes and may be beneficial to any future refinements of BlueCAM and other accounting frameworks.

The SEEA approach for sequestration does not incorporate avoided emissions and therefore is not reflective of the overall carbon abatement of the restoration project. For detailed accounts of the overall carbon abatement outcomes, see the BlueCAM derived accounts in the current section.

Uncertainties

There are multiple sources of uncertainty associated with generating carbon accounts for coastal wetland restoration projects, the most significant of which are described above in key assumptions and limitations. In most circumstances, it will be difficult to explicitly quantify the magnitude of these uncertainties for a specific project. There are, however, several pieces of information which provide some context to the expected uncertainty of carbon accounting approaches described here.

First, a sensitivity analysis was undertaken as part of the original development of BlueCAM (Lovelock et al. 2022). That is, 1000 Monte Carlo simulations of carbon abatement were run utilising 225 different combinations of BlueCAM baseline land uses (nine land uses), restored coastal wetland classes (five ecosystem types) and different climate regions (five regions) for abatement over 25 years. This sensitivity analysis showed that BlueCAM outputs were similar to the 40th percentile of the simulated outputs (i.e. BlueCAM estimates were lower than median simulation values) and are therefore conservative estimates of overall abatement.

Second, the application of carbon accounts described in this guide to the case studies of East Trinity Inlet and Tomago are informative of the relative uncertainty associated with choice of accounting level of detail (i.e. using nationally-consistent versus setting-specific approaches). In both case studies the overall abatement estimate was substantially higher under the setting-specific approach: 74 % higher in the case of East Trinity Inlet, and 100 % higher in the case of Tomago. This represents approach-level uncertainties in the order of 40,000 t CO_2e.over 20-years at East Trinity Inlet, and 7,400 t CO_2e.over 15-years at Tomago. Together, all these examples demonstrate that the use of BlueCAM without setting-specific inputs is more likely to lead to an underestimate of carbon abatement, as opposed to an overestimate of carbon abatement, for a given location. Discrepancies between the BlueCAM-derived estimates and SEEA-based carbon accounts are primarily due to the inclusion of avoided emissions under BlueCAM, and exclusion under SEEA, rather than differences in the uncertainty surround carbon parameter estimates (as both approaches utilise BlueCAM parameters as inputs).

[127] Lovelock, C. E., et al. (2022). Modeled approaches to estimating blue carbon accumulation with mangrove restoration to support a blue carbon accounting method for Australia. Limnology and Oceanography, 67:S50-S60. https://doi.org/10.1002/lno.12014

Table 7.5: Summary of BlueCAM calculator estimation parameters which may be repopulated with site-specific values where appropriate. Note the units required for each parameter, and any requirement for treatment of base values collected from the literature (which will typically be reported in units per unit of area per unit of time) prior to entry into the BlueCAM calculator worksheet.

BlueCAM estimation parameter	BlueCAM calculator row	Multiply base value by CEA area (ha)	Multiply base value by reporting timeframe (years)
Baseline avoided emissions of CO_2 ($E_{B,CO2}$) (tonnes CO_2)	V	yes	yes
Baseline avoided emissions of CH_4 ($E_{B,CH4}$) (tonnes CO_2e)	W	yes	yes
Baseline avoided emissions of N_2O ($E_{B,N2O}$) (tonnes CO_2e)	X	yes	yes
Coastal wetland emissions ($E_{CW,CO2}$) (tonnes CO_2e)	Y	yes	yes
Coastal wetland emissions ($E_{CW,CH4}$) (tonnes CO_2e)	Z	yes	yes
Coastal wetland emissions ($E_{CW,N2O}$) (tonnes CO_2e)	AA	yes	yes
ETR emissions from baseline vegetation (40 % AGB for 1 year) (t CO_2e ha^{-1})	BB	yes	no
ETR emissions from blue C wetlands (40 % AGB for 1 year) (t CO_2e ha^{-1})	BC	yes	no
AGB (t CO_2e ha^{-1})	BI	no	yes
BGB (t CO_2e ha-^{1})	BJ	no	yes
Soil C accumulation (t CO_2e ha^{-1})	BK	no	yes
Soil C stock (t CO_2e ha^{-1})	BL	no	yes

Table 7.6: Summary of approaches required to derive stock accounts for each CEA, utilising two complimentary model runs of BlueCAM calculator.

Stock account	BlueCAM input file(s) used	Equation applied to BlueCAM file for each CEA[1]
Aboveground biomass carbon stock (2007) [$\mathbf{AGB}_{baseline}$]	BlueCAM calculator Run #2 (Baseline Stocks) Note: all 'New CEA or first reporting period?' inputs to be entered as 'Yes'	$\mathbf{AGB}_{baseline} = BBx*2.5*Kx$ [Eqn 1]
Soil carbon stock to 1m (2007) [$\mathbf{Soil}_{baseline}$]	BlueCAM calculator Run #2 (Baseline Stocks) Note: all 'New CEA or first reporting period?' inputs to be entered as 'Yes'	$\mathbf{Soil}_{baseline} = BLx*Kx$ [Eqn 2]
Aboveground biomass carbon stock (2022) [$\mathbf{AGB}_{project}$]	BlueCAM calculator Run #1 (Abatement and Project Stocks) Note: $AGB_{baseline}$ parameter populated from Eqn 1 above	$\mathbf{AGB}_{project} = \mathbf{AGB}_{baseline} + BIx*Kx-ABx$ [Eqn 3]
Soil carbon stock to 1m (2022) [$\mathbf{Soil}_{project}$]	BlueCAM calculator Run #1 (Abatement and Project Stocks) Note: $Soil_{baseline}$ parameter populated from Eqn 2 above	$\mathbf{Soil}_{project} = \mathbf{Soil}_{baseline} + AMx$ [Eqn 4]

[1] where italicised letters (e.g. *K*, *AB*, *BB*) refer to BlueCAM Excel columns and 'x' refers to the Excel row number for a given CEA)

Table 7.7: Template for BlueCAM output values for each Carbon Estimation Area (CEA) derived from classification of land type changes as determined from either nationally-consistent or detailed approaches for a restoration project.

			Carbon abatement			Carbon stocks			
CEA ID	**CEA baseline land type**	**Land type for CEA: current reporting period end**	**CEA total emissions avoided ($E_{A,i}$) (tonnes CO_2e)**	**CEA total carbon sequestered in vegetation ($C_{V,i}$) (tonnes CO_2e)**	**CEA total carbon sequestered in soil (tonnes CO_2e)**	**Vegetation biomass carbon stocks - baseline AGB tCO_2e**	**Vegetation biomass carbon stocks - project AGB tCO_2e**	**Soil carbon stocks - baseline tCO_2e**	**Soil carbon stocks - project tCO_2e**
			BlueCAM [sheet] and column:			*BlueCAM [sheet] and column:*			
CEA1	[baseline land type 1]	[reporting period land type 1	[Run #1] AC	[Run #1] AG	[Run #1] AM	[Run #2] New column*	[Run #1] New column*	[Run #2] New column*	[Run #1] New column*
CEA2	[baseline land type 1]	[reporting period land type 2]							
CEA3	[baseline land type 1]	[reporting period land type 3]							
CEA4	[baseline land type 2]	[reporting period land type 1]							
CEA5	[baseline land type 2]	[reporting period land type 2]							
etc.									
TOTAL			**sum of all CEAs**	**sum of all CEAs**	**sum of all CEAs**	**sum of all CEAs**	**sum of all CEAs**	**sum of all CEAs**	**sum of all CEAs**

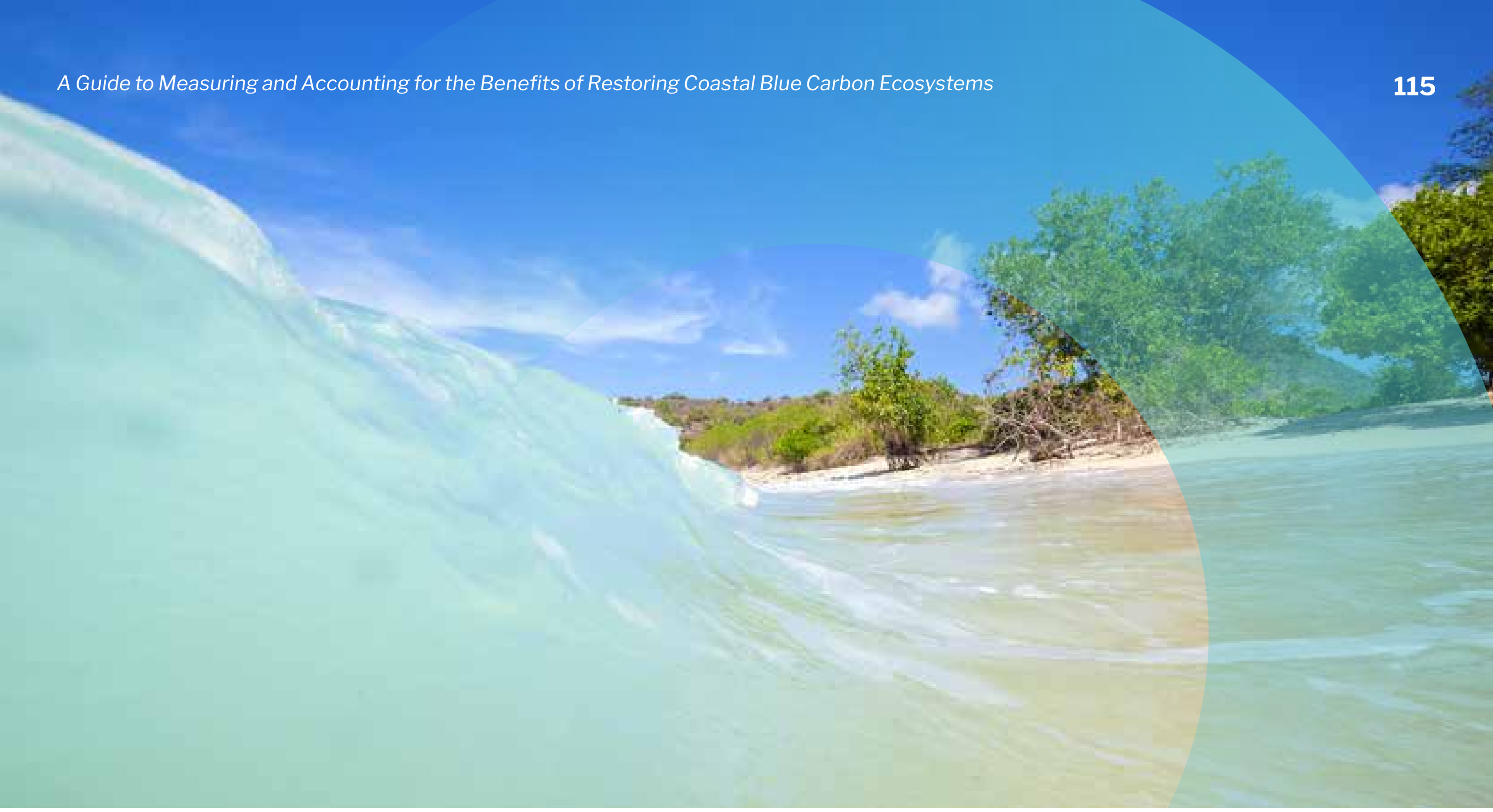
* these columns are not supplied with the generic version of the BlueCAM calculator and need to be generated as per the equations provided in Table 7.6.

Table 7.8: **Template for BlueCAM-derived physical accounts, and financial accounts of carbon abatement as applied to both nationally-consistent or detailed approaches.** Australian Carbon Credit Unit (ACCU) value of \$30.75 is based upon September 2022 spot price, and 'premium' price based on an arbitrary value of \$150 per unit.

	(A) Nationally-consistent approach or (B) Detailed approach		
BlueCAM outputs:	**Physical account (Tonnes CO_2e)**	**Financial account (AUD) - ACCU SPOT**	**Financial account (AUD) - Premium**
1. Reporting period emissions avoided (E_A)	[BlueCAM output]	E_A x 30.75	E_A x 150.00
2. Reporting period C sequestered in vegetation and soil (CP)	[BlueCAM output]	CP x 30.75	CP x 150.00
3. Emissions from fuel consumed during reporting period (E_{fk})	[BlueCAM output]	E_{fk} x 30.75	Ef_k x 150.00
4. Net abatement amount (A_r): BlueCAM calculator output	[BlueCAM output]	A_r x 30.75	A_r x 150.00
Net abatement amount ($A_{r\text{-}adj}$): ERF discount removed	**Sum of E_A + CP + E_{fk}**	**$A_{r\text{-}adj}$ x 30.75**	**$A_{r\text{-}adj}$ x 150.00**

Table 7.9: **Template for BlueCAM-derived physical accounts, and financial accounts of carbon abatement as applied to both nationally-consistent or detailed approaches.** Australian Carbon Credit Unit (ACCU) value of \$30.75 is based upon September 2022 spot price, and 'premium' price based on an arbitrary value of \$150 per unit.

	(A) Nationally-consistent approach or (B) Detailed approach
BlueCAM outputs:	**Physical account (Tonnes CO_2e)**
1. Vegetation aboveground biomass carbon stocks – baseline	[Table 7.7 'sum of all CEAs']
2. Vegetation aboveground biomass carbon stocks – project	[Table 7.7 'sum of all CEAs']
3. Soil carbon stocks – baseline	[Table 7.7 'sum of all CEAs']
4. Soil carbon stocks - project	[Table 7.7 'sum of all CEAs']
Total carbon stocks – baseline	[sum of row 1 + 3]
Total carbon stocks – project	[sum of row 2 + 4]
Net carbon stock change (project - baseline)	**[difference of above]**



8. Water purification services

Dr Maria Fernanda Adame

8.1 Summary of section

Wetlands can significantly improve the quality of the water that inundates them. Thus, many restoration projects worldwide have targeted wetland restoration to reduce water pollution. Wetlands can improve water quality by reducing nutrients (nitrogen, N, phosphorus, P) and total suspended solids (TSS) from the water column[128]. N can be removed through denitrification, the process in which soil microorganisms convert nitrate (NO_3^-) to gaseous N_2. Trees can also remove N (primarily as ammonia, NH_4^+) and dissolved P (e.g. as phosphates PO_4^-) and store it as wood. Additionally, sediment accretion can retain total suspended solids, particulate N and P in the wetlands (Figure 8.1).

In cases where wetland restoration objectives include remediating acid sulphate soils, tidal flushing will restore natural acidity values in the water and sediment. Thus, to quantify the values of the restoration of a coastal wetland for improving water quality, four main processes should be considered: 1) denitrification, 2) tree uptake, 3) sediment accretion, and 4) pH regulation.

[128] Land, M. et al. 2016. "How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review" (2016) Environmental Evidence 5:9

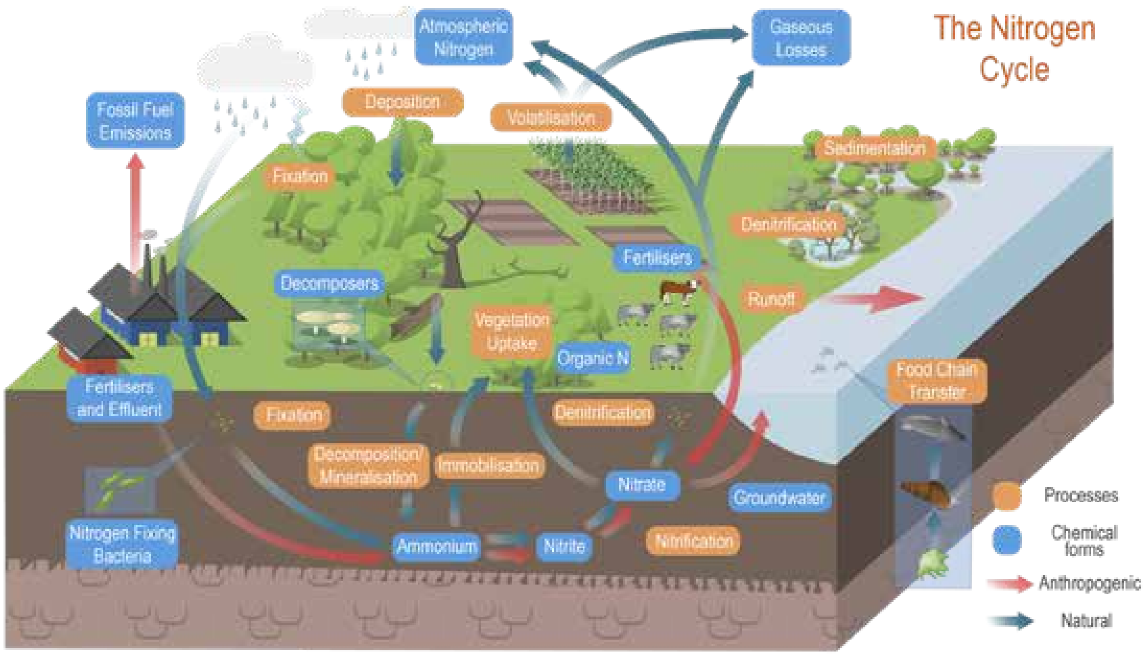


Figure 8.1: Nitrogen conceptual model and the role of coastal wetlands in denitrification and sediment accretion for nitrogen removal[129]

8.2 Detailed section

Denitrification

Denitrification is the primary process responsible for water purification in wetlands. Anoxic, carbon-rich soils in coastal wetlands are ideal for microorganisms to transform NO_3^- to NO_2^- and eventually to gaseous N_2, permanently removing nitrogen from the water column (Figure 8.2).

A relationship between NO_3^- concentrations and denitrification has been established for wetlands globally and in northeast Australia[130] (Figure 8.3). This relationship can be used to determine the removal potential of NO_3^- per hour of inundation (mg/m^2/hr).

To obtain the denitrification potential for a restored wetland it is necessary to know the concentrations of nutrients that are in contact with the soil and the frequency of inundation. Water quality datasets are necessary to obtain this information. These are available at the State and regional scale. For instance, long-term state monitoring programs include:

- QLD: https://www.qld.gov.au/environment/water/quality/monitoring
- NSW: https://www.environment.nsw.gov.au/topics/water/water-quality/monitoring-and-reporting
- VIC: https://data.water.vic.gov.au/
- WA: https://www.water.wa.gov.au/water-topics/waterways/threats-to-our-waterways/statewide-river-assessment
- TAS: https://nre.tas.gov.au/water/water-monitoring-and-assessment/water-monitoring/surface-water-quality/water-parameters

[129] Department of Environment and Science, Queensland (2021) Nitrogen processes, WetlandInfo website, accessed 2 June 2023. Available at: https://wetlandinfo.des.qld.gov.au/wetlands/ecology/processes-systems/nitrogen-concept-model/

[130] See Piña-Ochoa, E., & Álvarez-Cobelas, M. (2006). Denitrification in aquatic environments: a cross-system analysis. Biogeochemistry, 81:111–130. https://doi.org/10.1007/s10533-006-9033-7; OR Adame, M. F., et al. (2019). Tropical coastal wetlands ameliorate nitrogen export during floods. Frontiers in Marine Science, 6, 671. https://doi.org/10.3389/fmars.2019.00671

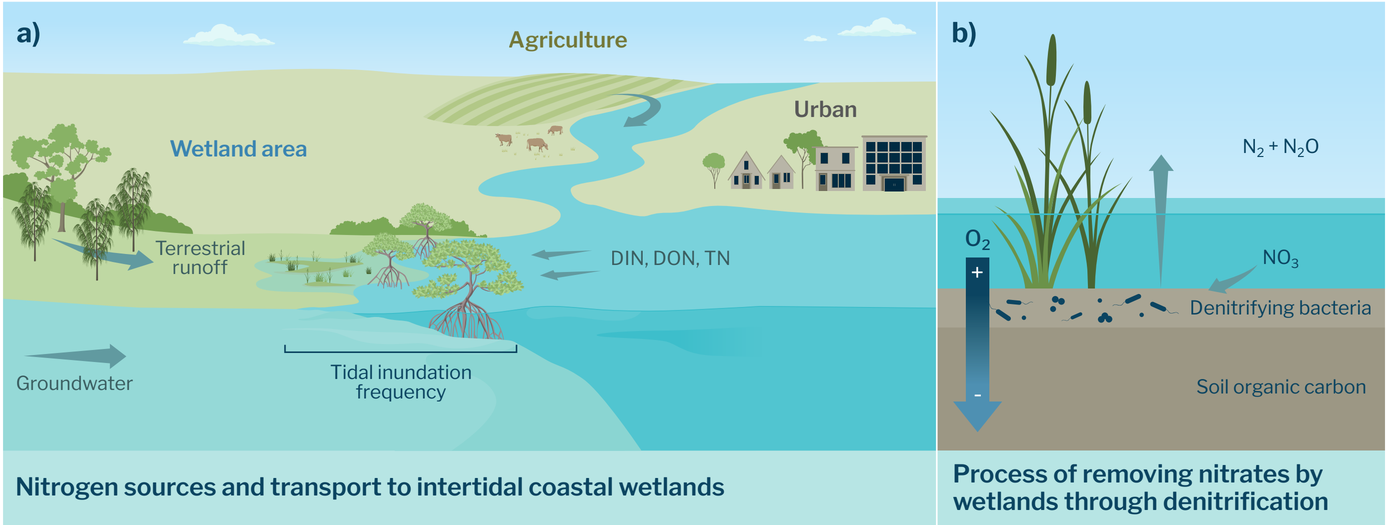


Figure 8.2: a) Nitrogen moving from terrestrial sources through rivers, surface runoff, and groundwater to coastal wetlands as total (TN), dissolved inorganic (DIN= NO_3^-+ NH_4^+) and organic nitrogen (DON), b) wetlands remove NO_3^- through denitrification by microorganisms in waterlogged sediments with low oxygen

There are also regional monitoring programs, for instance:

- Great Barrier Reef water quality program: https://www2.gbrmpa.gov.au/our-work/programs-and-projects/marine-monitoring-program
- And in most coastal cities of Australia, for example, Darwin, NT: https://depws.nt.gov.au/water/water-management/darwin-harbour/Darwin-Harbour-water-monitoring

When selecting a dataset, it is crucial to consider that nutrient concentrations are highly temporarily variable (and not normally distributed), with high peaks usually found after rainfall events. Thus, datasets that include seasonal and interannual variations (at least two years) are preferred. The dataset must be as close to the site as possible and should be the primary source of nutrients into the wetland. For instance, a mangrove forest in the mouth of the river should use data sets close to the river mouth, while a mangrove forest that only is flooded by tidal water should only include datasets from oceanic waters.

In some restoration projects monitoring of in versus outflows is available, and this setting would be ideal to monitor N, P and TSS removal. Guidelines for some states, such as Queensland, are available to set, sample and analyse data of nutrient removal (see Department of Agriculture and Fisheries). However, many restoration projects will not have the funding to continuously monitor nutrient and sediment uptake, in this case, the following estimations can be conducted:

Monthly or seasonal NO_3^- concentrations from water source into the wetlands can be used to obtain denitrification potential (mg/m²/h) following the relationship in Figure 8.3 and Equation 1.

$$\log(\text{denitrification, mg } m^2 h^{-1}) =$$

$$0.5093(\log((NO^3(\text{mg } L^{-1}) + 3)) + 0.174$$

Equation 1

Denitrification requires soils that are rich in organic carbon. Thus, denitrification potential should only be calculated for vegetated areas, i.e., mangroves and saltmarsh, but not for unvegetated saltpans, mudflats, or lagoons[131]. Once the potential for denitrification has been established, NO_3^- removal can be estimated from vegetation area (in ha) and inundation frequency (hr/yr). If concentrations of N are only available as dissolved inorganic nitrogen (DIN = NO_3^- + NH_4^+). These can be used if the proportion of NO_3^- is known (e.g. 90 % DIN is NO_3^-).

[131] Adame M.F., et al. (2021). Denitrification within the sediments and epiphyton of tropical macrophyte stands. Inland Waters ,11:257–266. https://doi.org/10.1080/20442041.2021.1902214

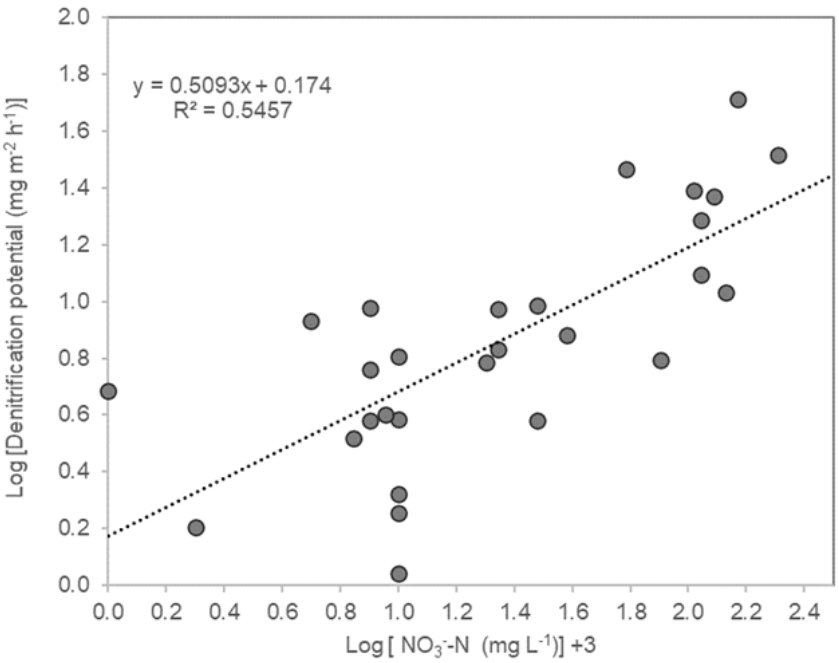


Figure 8.3: Relationship between [log] denitrification potential (mg m^{-2} h^{-1}) and NO_3^- concentration [log+3] (mg L^{-1}) for coastal wetlands in North Queensland. The regression is significant at R2 = 0.545; F 1, 27= 31.23, $p < 0.001$.

Vegetation area can be obtained from the extent section. Frequency of inundation is more challenging to obtain, and in situ loggers of water depth combined with bathymetry maps are the best option. If funds are unavailable for a complete hydrological analysis, there are two other options. The first is to obtain inundation frequency from satellite images, such as Water Observations from Space (WOFS: https://cmi.ga.gov.au/catalog/dea-water-observations-wofs; Figure 8.4).

However, these products are only useful in saltmarsh, as in forested coastal wetlands (mangroves and supratidal forests), tree canopy masks the surface water signal, incorrectly suggesting lower inundation where canopy cover is high (the opposite is also true).

The third option is to assume tidal inundation frequency depending on vegetation type. Vegetation composition in coastal wetlands is stratified according to elevation and, thus, tidal inundation frequency (Figure 8.5). Mangroves are usually flooded at least once daily, while supratidal forests are only flooded during large tides or by rainfall and runoff during wet periods.

Thus, we can assume that the lowest in the intertidal, the more frequently the vegetation type will be inundated. To estimate the frequency of inundation, tidal height predictions can be obtained from the closest tide monitoring station (e.g. Bureau of Meteorology, http://www.bom.gov.au/oceanography/projects/ntc/tide_tables.shtml). It is possible to assume that tidal inundation would occur for mangroves only when tides are above the 50th percentile, for saltmarsh when they are above the 75th percentile and for supratidal wetlands, only during the highest tides, at the 90th percentile (see example in Table 8.1). The assumptions should be corroborated with field knowledge from the site.

There are also local hydrological factors to consider. For instance, some coastal wetlands are affected by land runoff, artificial drains, or groundwater intrusion. These are likely to be sources high in nutrients, and they should be considered when estimating the water quality benefit for each ecosystem.

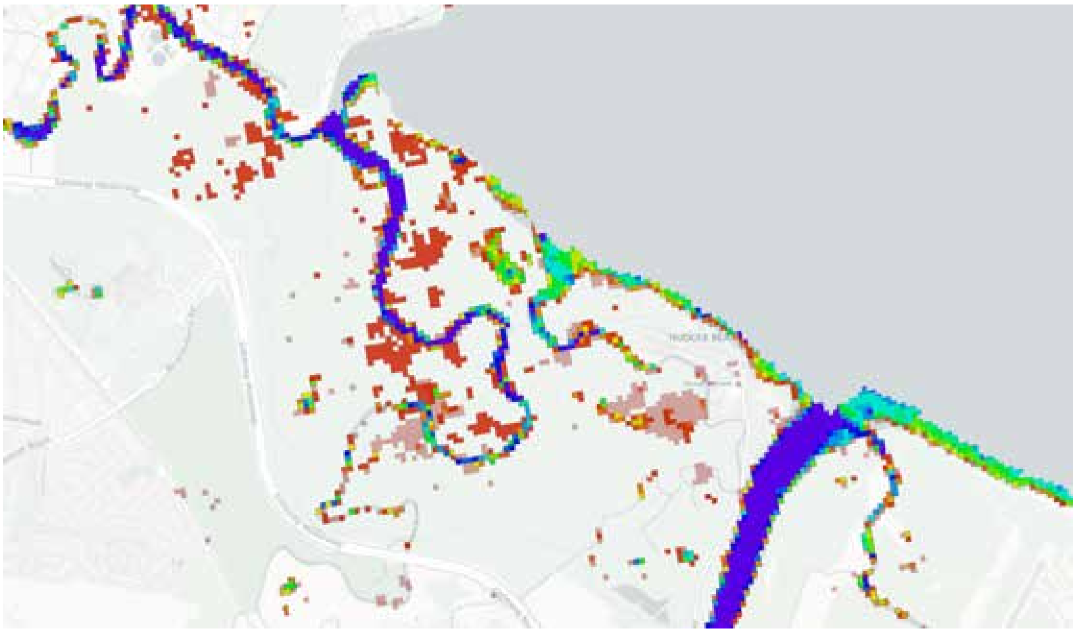


Figure 8.4: Example of annual water inundation frequency in Nudgee wetlands Brisbane, Queensland, where areas in red show where water inundation was observed 1% of the time (i.e. 36 days), and blue areas, where water was observed 100% of the time (365 days). Data is from Digital Earth Australia, Water Observations from space (https://maps.dea.ga.gov.au/)

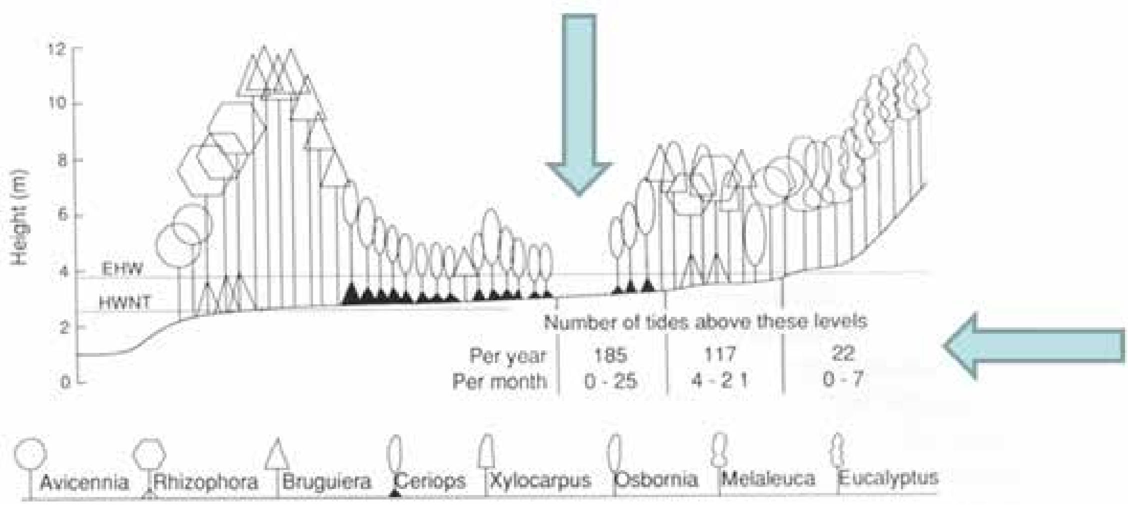


Figure 8.5: Inundation frequency versus vegetation type[132].

[132] Source: McNae, W. 1963. Mangrove swamps in South Africa. Journal of Ecology 51: 1-25.

Table 8.1: Characteristics of the tidal regime in Cairns (ABM, annual tide predictions for 2024): height (m) and number of tides at that height per year. HAT = highest astronomical tide

	(m)	No tides/yr
HAT	1.74	
Maximum	3.57	
Minimum	0.07	
25th	1.07	1078
50th	1.65	716
75th	2.45	349
90th	2.81	167

Tree uptake and storage as woody biomass

Trees can remove nutrients such as N and P and store them in the long-term by trees as woody biomass. Nutrient uptake by trees can be estimated from growth rates calculating changes in biomass with time[133] (kg/tree/yr). The biomass accumulation is then converted to N and P from their concentrations in wood (kgN or kgP/tree/yr) and then converted to an area (kgN/ha) from estimations of forest density (trees/ha)[134]. Wood N and P concentrations can be obtained in the field by extracting wood cores with a borer (Figure 8.6) or using published data values. For instance, Melaleuca spp. trees in Northern Queensland have N concentrations of 0.1 %[104], while mangroves of Rhizophora apiculata have N concentrations of 0.5 %[135].

Soil sequestration of nutrients and suspended solids

N removal can be estimated from soil carbon sequestration rates[136] and converted to N and P with information of C:N:P of surface soils. Alternatively, it can be estimated from sediment accretion rates (mm/yr) and bulk density (g/cm³) or from the relationship between turbidity and total sediment retention[137] (Equation 2). The turbidity of the water that inundates the wetland can be obtained from some water quality datasets, but it is not always reported.

$$\textit{Surface accretion (mm/month)} = 0.261 + 0.016\ \textit{(Turbidity, NTU)}$$

Equation 2

Surface accretion can then be transformed to aerial accumulation per year using soil bulk density values from the site and then converted to kg of suspended sediments per hectare of wetland. The N and P accumulated can be obtained if nutrient content of the suspended or deposited sediment is known.

133 Komiyama, A., et al. (2008). Allometry, biomass, and productivity of mangrove forests: A review. Aquatic Botany 89, 128–137.

134 Adame M.F., et al..(2019). Carbon and nitrogen sequestration of Melaleuca floodplain wetlands in tropical Australia”. Ecosystems 23. 454–466

135 Gong, W.-K. K. & Ong, J. E. E. (1990). Plant biomass and nutrient flux in a managed mangrove forest in Malaysia. Estuarine, Coastal and Shelf Science. 31:519–530.

136 Serrano O., et al. (2019).Australian vegetated coastal ecosystems as global hotspots for climate change mitigation”. Nature Communications. 10,4313.

137 Lovelock C.E., et al. (2014). Contemporary rates of carbon sequestration through vertical accretion of sediments in mangrove forests and saltmarshes of South East Queensland, Australia. Estuaries and Coasts 37:763–771. https://doi.org/10.1007/s12237-013-9702-4

Figure 8.6: Sampling for wood cores from a Melaleuca tree to obtain N and P concentrations, Insulator Ck, Qld. Picture M. F. Adame.

Acidity reduction

Changes in sediment and pH values as the result of the restoration can be obtained from water quality monitoring programs, which are usually required when performing activities in acid sulphate soils. The changes (pre- and post-) in pH and soil acidity (mol H+/t) can be established as the result of the restoration activity.

For example, in Trinity Inlet a monitoring program was established to monitor changes in water acidity (as H+ mol/L and as pH) before and after the restoration of a wetland with activities that included lime application since 2009. The results show how the activity improved the acidity levels of the water (Figure 8.7). The changes in

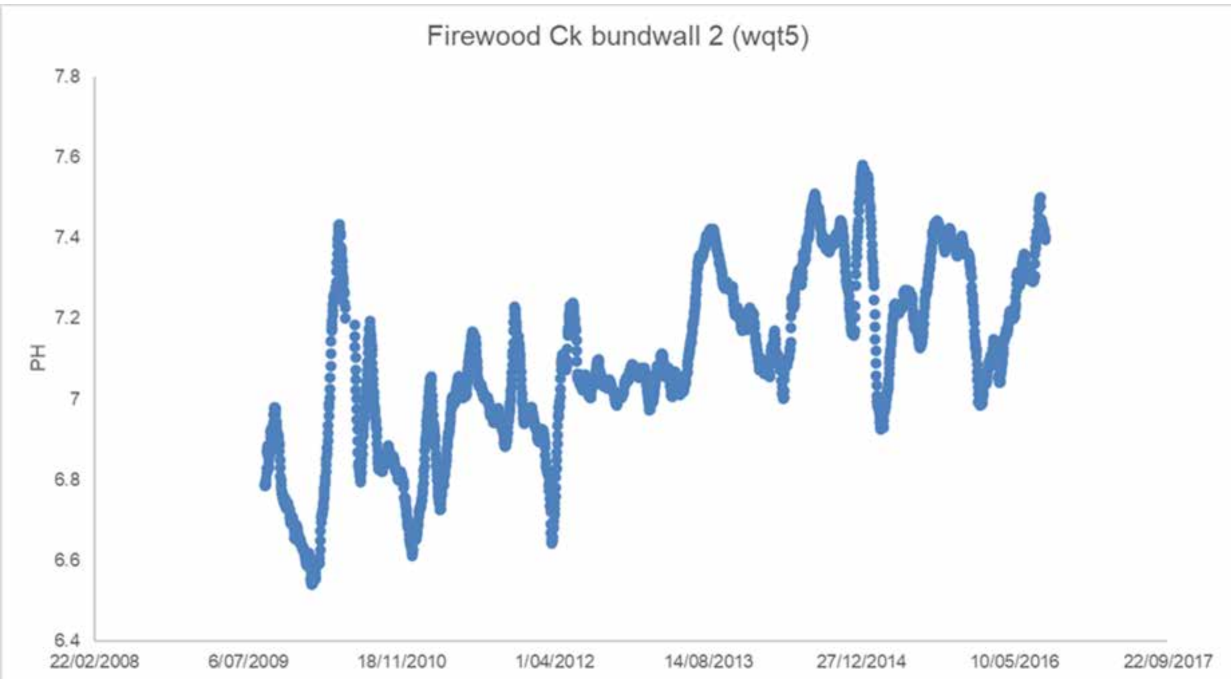


Figure 8.7: Water pH from Firewood Creek bundwall monitoring station from 2009 to 2016. Data is from the Department of Environment and Sciences.

acidity can be used as a measure of water quality improvement.

This guide provides three standards of quality/cost for estimating the processes associated with improving water quality from coastal wetland restoration. These are Gold standard for high-quality/cost, the Silver standard for mid-quality/cost, and Bronze standard for the lowest quality/cost option (Table 8.2). In reality, a restoration project is likely to use a combination of these depending on budget and data availability for each component.

Key assumptions or limitations

The largest source of error from the estimations of improvement of water quality are:

- **Seasonal or interannual variations of nutrient concentration**

  Nutrient concentrations are highly temporarily variable, with concentrations usually peaking after rainfall events. Dissolved nutrients are also not normally distributed, meaning that infrequent sampling may incorrectly assume mean values. This source of error can be overcome by selecting long-term, high-frequency water quality datasets. Additionally, conducting at least some in situ sampling of dissolved nutrients during dry and wet periods can significantly improve estimations. The sampling of dissolved nutrients is relatively straightforward with many laboratories offering analyses of dissolved inorganic nutrients at affordable prices.

- **Vegetation area**

  Spatial analyses of vegetation area can be very useful in determining vegetation types, mangroves are especially easy to identify as they are evergreen forests. However, other forest types such as *Melaleuca* and *Casuarina* forests are more difficult to identify from spatial analyses. Field verifications of vegetation types are crucial to avoid mis-identifying vegetation groups from spatial datasets.

- **Inundation frequency**

  This error is likely to be the largest assumption from the estimation of water quality improvement. Currently, there is not a spatial product that can unequivocally distinguish water from tree canopy, thus images tend to show lower inundation where trees are highest and vegetation is dense, which in fact, the opposite is true. The best option to determine inundation frequency is to conduct a hydrological study of the site. However, these studies are expensive and technologically complex.

The best approach when information is lacking or unreliable is to assume different inundation frequencies depending on vegetation type, which is a well-founded ecological principle in coastal wetlands. However, to define the limit at which tidal amplitude each vegetation type is inundated, local information is essential (e.g. only tides > 3.5 m inundate the saltmarsh).

Monetary valuation

Many locations around Australia have specific programs to limit nitrogen input into marine and coastal waters (e.g. Great Barrier Reef, Southeast Queensland, and Port Phillip Bay, Victoria). In these instances, there is a price put on Nitrogen (as in dissolved inorganic nitrogen, DIN or total nitrogen, TN) as an offset through some activities. These include gully restoration and wetland restoration/creation (Reef Credits, Great Barrier Reef), or wetland restoration for offsetting N from treated water (e.g. Southeast Queensland). The price is typically placed on 1 kg of N, and can be directly applied to the amount of Nitrogen removed from the system calculated above (kg N yr^{-1}). In locations where Nitrogen is not an issue, and there is no direct price on it, then it may not be relevant to assess the water quality improvements beyond that considered in the condition accounts (Section 6).

Uncertainties

The datasets used to determine the water quality benefits have a known level of uncertainty. For instance, water quality detections are usually > 0.001 mg/L, and spatial imagery is limited by resolution (e.g. 10 m pixels). To facilitate the visualisation of the uncertainty for each parameter, mean, median, standard error and ranges of values should be incorporated in the results if possible. The largest source of variability is not the water sampling itself (~0.01 mg/L) but the impacts of periods of rainfall, which increase variability by orders of magnitude (e.g. 0.01 to 1 mg/L). This variability should considered when calculating an annual nutrient uptake budget, especially if most sampling occurred during or shortly after periods of rainfall. The second largest source of uncertainty is that of changes in wetland area, as most of the changes in water quality are driven by area. Thus, a sensitivity analysis that incorporates a range of scenarios with different areas within the spatial image resolution (± 10 m^2) should be included to better represent the effects of uncertainty.

Table 8.2: Estimation of water quality improvement through denitrification, tree uptake and sedimentation from the restoration of coastal wetlands with High, Medium and Low standards. project.

	Known - High confidence Cost > $100K	Derived - Medium - confidence $50 - 100 K	Derived - Low - confidence $20 - 50 K
Denitrification			
NO_3^- concentration	*In situ* monitoring of water flows: in and out of the wetland. Or in situ monitoring of NO_3^- concentrations in the inflow including seasonal and interannual variations	NO_3^- concentrations from nearby monitoring programs, including seasonal and interannual variations, with some values confirmed on site.	NO_3^- concentrations from the closest monitoring program
Vegetation cover	*In situ* monitoring of vegetation species and cover	Spatial images of vegetation types and cover confirmed by field data.	Spatial images of vegetation types and cover
Inundation frequency	*In situ* bathymetry data and high-frequency loggers of inundation depth. Or, regular monitoring of water flows.	Spatial images of water changes verified with some field data of inundation depth and frequency	Assumptions of tidal inundation frequency depending on vegetation type
Tree uptake			
Tree growth as biomass	*In situ* measurements of growth rates and tree density	Growth rates and forest density of studies in nearby coastal wetlands verified with some field data	Growth rates and forest density of studies in similar coastal wetlands
Forest density	*In situ* measurements of forest density	Forest density from studies in nearby coastal wetlands verified with some field data	Forest density from studies in similar coastal wetlands
Wood N and P	*In situ* analyses of wood N and P	Wood N and P from nearby coastal wetlands	Wood N and P from similar coastal wetlands
Sediment accretion/sequestration			
Sedimentation/ accretion rates	*In situ* monitoring of short-term sediment accretion and long-term sequestration	Accretion and long-term sequestration of nearby coastal wetlands	Accretion and long-term sequestration of similar coastal wetlands
Sediment N and P content	*In situ* sampling of N and P of suspended and deposited sediments	N and P concentrations suspended and deposited in nearby coastal wetlands	N and P concentrations suspended and deposited in similar coastal wetlands
Decrease in acidity			
Changes in acidity (soil and water)	*In situ* measurements of water acidity (H+, mol/L) before and after restoration	*In situ* measurements of water pH before and after restoration	*In situ* measurements of pH after restoration to show soil and water neutrality

9. Coastal protection: erosion, storm mitigation, flood control services

Prof. Will Glamore, Alice Harrison

9.1 Summary of section

Coastal regions in Australia are susceptible to damage from floods, erosion and storms due to their low-lying nature and position in the coastal zone. Erosion reduction, storm mitigation, and flood control ecosystem services are grouped together as these services buffer natural processes (including natural disasters) and can reduce the damage to human infrastructure. Blue carbon ecosystems provide these services through a variety of mechanisms, primarily via energy adsorption (e.g. wave attenuation, frictional drag or water storage), which decreases the quantity or severity of assets at risk. The value of these services is spatially variable, due to the geographic variability of climate/bio-physical processes (the threat), vegetation characteristic (e.g. density or width of vegetation) (the service), and human infrastructure (the benefit)[138].

While the three ecosystem services considered in this section are related, they are distinct and may not apply at every restoration site. Further, natural hazards can either be event-based, such as localised flooding or cyclonic storm surges, or associated with long-term trends, such as channel migration. While methods for quantifying these services may be similar, the processes and datasets are sufficiently different. As such, this section is split into three services:

- Long term, persistent erosion processes (i.e. not event based)
- Flood mitigation services, associated with riverine flooding (extreme event based)
- Coastal inundation associated with storm tides, particularly coastal cyclonic events (extreme event based)

[138] Crossman, N. D., et al. Brander, L. (2019). Water flow regulation for mitigating river and coastal flooding. SEEA EEA Revision 15

Long term hazards:

- **Persistent erosion (i.e., not event based) processes** – Some estuarine and marine environments display long-term recessionary trends due to the effects of high flows (at velocities sufficient to mobilise sediments) or waves (including swell waves, wind waves or boat wakes). Coastal wetlands can reduce bank or dune recession through various mechanisms, including wave or flow attenuation or by naturally armouring the foreshore. Blue carbon ecosystems will primarily provide a protection benefit to areas/infrastructure immediately adjacent to the restored site.

Moderation of extreme events:

- **Flood mitigation services, associated with riverine flooding** – these services are associated with the reduction of catchment flooding impacts on human infrastructure, as illustrated in Crossman et al. (2019)[139] separates the functional role of reducing flood impacts into two categories: prevention and mitigation. Prevention functions are associated with terrestrial ecosystems, such as upland forests, which can reduce runoff in a flood event by intercepting waters in the catchment. This preventative service is not considered relevant for blue carbon ecosystems, which are associated with high water tables due to the presence of tides and are located in the lower reaches of a catchment. Alternatively, mitigation functions are commonly associated with wetlands and coastal floodplains, recognising the storage of surplus water during a flood event, reducing flood peaks, and altering flood inundation times throughout the floodplain. This mitigation function may be significant for some blue carbon restoration sites, particularly where they represent a large proportion of the coastal floodplain area within an estuary. Importantly, flood mitigation services are most likely derived from increased flood storage volume in the wetlands after the removal of infrastructure such as levees or floodgates, rather than the slowing of water movement due to the presence of vegetation. In the case of restored areas, coastal wetlands also increase flood resilience if valuable assets (e.g. buildings or agricultural lands) are moved to higher land, reducing potential flood damage costs. Unlike the other services in this section, flood mitigation services can provide benefits to areas that are not directly next to the ecosystems and must be considered in a whole-of-floodplain approach.

- **Coastal inundation associated with storm tides, particularly coastal cyclonic events** – Storm tides are the combination of normal astronomical tides and storm surges. Storm surges are particularly damaging in areas affected by cyclonic events, and there is substantial evidence supporting the value coastal wetlands in reducing damages during tropical cyclones[140;141;142;143;144]. For instance, coastal

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[139] Crossman, N. D., et al. (2019). Water flow regulation for mitigating river and coastal flooding. SEEA EEA Revision 15

[140] Beck, M. W., et al. (2022). Return on investment for mangrove and reef flood protection. Ecosystem Services, 56:101440. https://doi.org/10.1016/j.ecoser.2022.101440

[141] Costanza, R., et al. (2008). The value of coastal wetlands for hurricane protection. Ambio, 37:241-248. https://doi.org/10.1579/0044-7447(2008)37[241:TVOCWF]2.0.CO;2

[142] Mulder, O.J., et al. (2020). The value of coastal wetlands for storm protection in Australia. Ecosystem Services, 46:101205. https://doi.org/10.1016/j.ecoser.2020.101205

[143] Narayan, S., et al. (2017) The value of coastal wetlands for flood damage reduction in the northeastern USA. Scientific reports, 7(1): 9463. https://doi.org/10.1038/s41598-017-09269-z AND Wamsley, T. V., et al. (2010). The potential of wetlands in reducing storm surge. Ocean Engineering, 37(1): 59-68. https://doi.org/10.1016/j.oceaneng.2009.07.018

[144] Wamsley, T. V., et al. (2010). The potential of wetlands in reducing storm surge. Ocean Engineering, 37(1): 59-68. https://doi.org/10.1016/j.oceaneng.2009.07.018

wetlands can reduce damages associated with storm tides by absorbing energy from the storm[145;146]; via increased drag forces on water (attenuating storm surge and associated coastal flooding), demonstrated in the conceptual diagram in Figure 9.1. Blue carbon ecosystems provide storm protection benefits by being a physical barrier and provide protection to adjacent areas. While storm tides often coincide with low pressure systems associated with heavy rainfall and riverine flooding, those processes are considered separately as part of flood mitigation services associated with riverine flooding.

Note that blue carbon ecosystems can also improve water quality associated with extreme events through the entrapment of sediment and nutrient removal. However, this is considered a water quality service and is not included in this section (see Section 8).

[145] Beck, M.W., et al. (2022). Return on investment for mangrove and reef flood protection. Ecosystem Services, 56:101440. https://doi.org/10.1016/j.ecoser.2022.101440

[146] Costanza, R., et al. (2008). The value of coastal wetlands for hurricane protection. Ambio, 37:241-248. https://doi.org/10.1579/0044-7447(2008)37[241:TVOCWF]2.0.CO;2

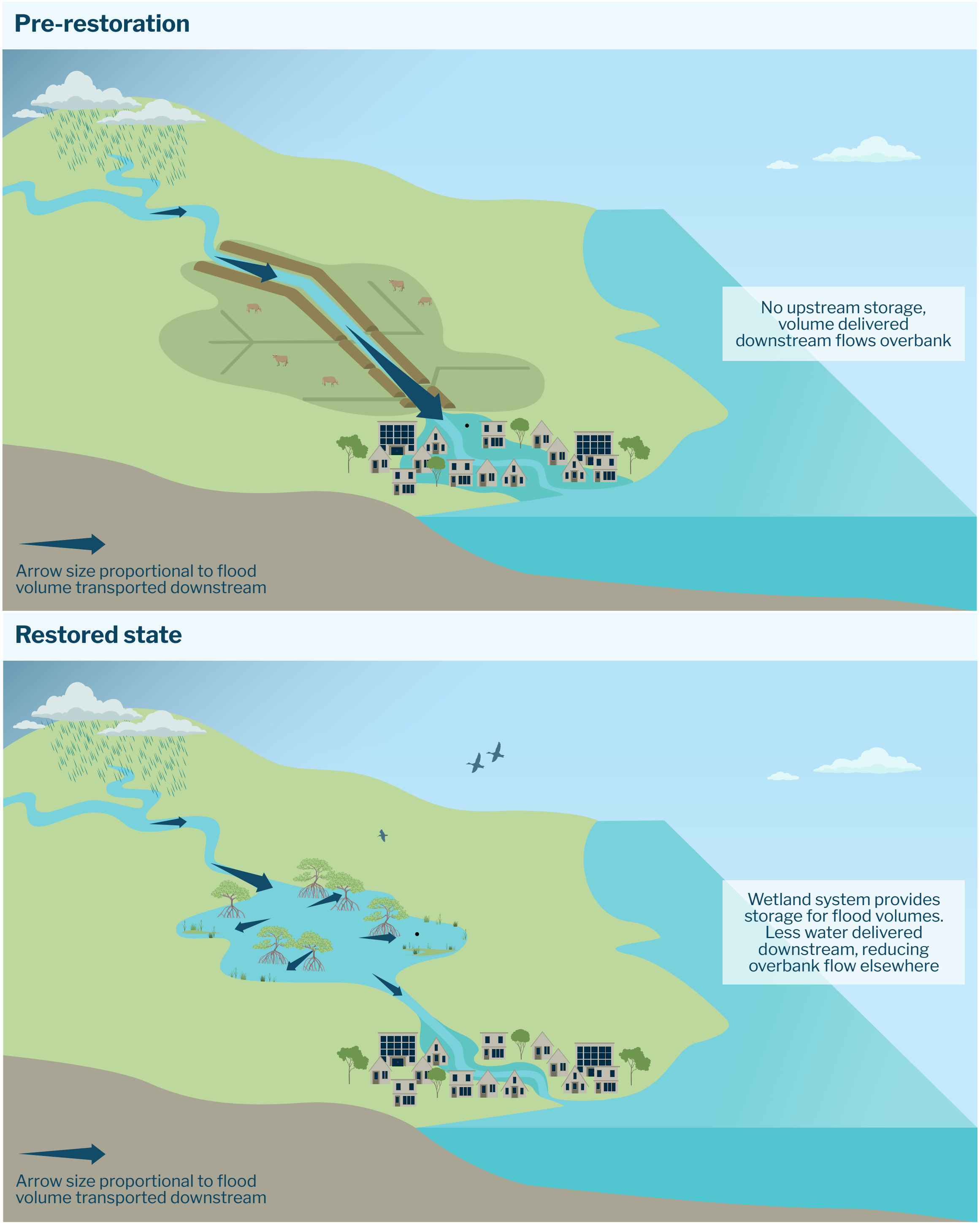

Figure 9.1: Conceptual diagram of flood mitigation associated with riverine flooding.

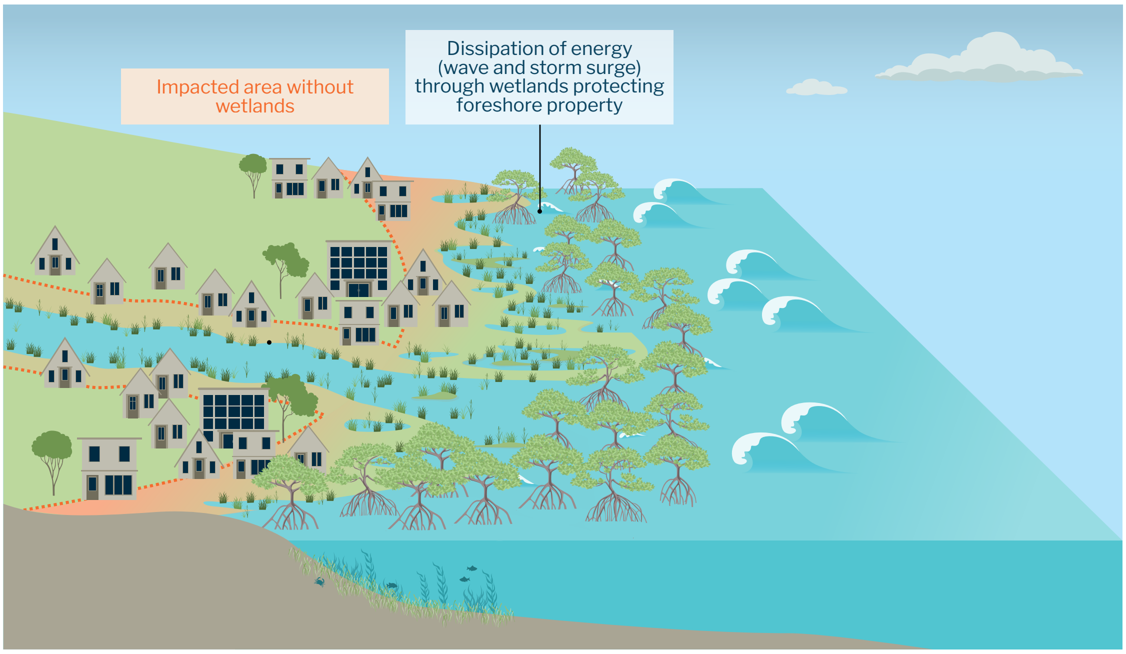

Figure 9.2: Conceptual diagram of coastal inundation associated with storm tides, particularly coastal cyclonic events.

9.2 Detailed section

Methods

As discussed above, the three services considered in this section are to be quantified using a five-step approach (Figure 9.3). The first three steps are a low investment screening process to rapidly assess the potential for these benefits to be relevant to a given site, and whether further resource heavy quantification should be considered. The final two steps provide guidance on quantifying the physical and monetary accounts where relevant. Each step is considered here in further detail.

Step 1: Do the relevant environmental hazards exist?

This step tests whether the environmental hazards are relevant to the site. This will typically be assessed based on historical evidence of hazards or the existence of hazard mapping.

Persistent erosion

Unlike the two hazards discussed in this section, persistent (or chronic) erosion is not associated with extreme events. Long-term recessionary trends of banks or dunes are often due to the regular effects of high flows (at velocities sufficient to mobilise sediments) or waves (including swell waves, wind waves or boat wakes). Critically, persistent erosion is associated with sustained removal of sediments from the foreshore. This is different from event-based erosion events on sandy coastlines where dune erosion occurs during large swell events, but it followed by accretionary periods restoring the beach to an 'average' state'. Persistent erosion should be evidenced by baseline monitoring records of recession. At this stage of the assessment, this may be through anecdotal evidence or observations, however more detailed quantification of the erosion is required for the accounts process, as described in later sections.

Riverine flooding

Riverine flooding occurs on floodplains where the flow conveyance capacity of waterways is exceeded during heavy rainfall, causing overbank water flows. While flood processes can occur almost everywhere, in most practical situations for this assessment, evidence of relevant flood processes will likely include:

- Existence of a flood study, showing water levels major waterways adjacent to the restoration area exceed the floodplain elevation, or
- Historical records of flood levels exceeding the elevation of the restoration area floodplain.

Coastal inundation associated with storm tides, particularly cyclonic events

Storm tide is a term used to refer to the water levels associated with a combination of storm surge (abnormal rise in ocean water levels above normal tides, typically associated with strong winds and low-pressure systems) and normal astronomical tides. While storm tides can occur in any coastal location, storm tides present a significant risk in areas affected by tropical cyclones, where storm surges can result in storm tide water levels several metres above normal tidal water levels. These areas are considered the most likely to have benefits in protection from coastal inundation from the restoration of coastal wetland ecosystems. Storm tides associated with cyclonic events occur in parts of Australia impacted by extreme low-pressure systems. Cyclones occur primarily in the tropical northern coastal regions of Australia. The cyclone affected areas in Australia are defined by the wind regions in the Australian Standards (AS/NZS 1170.2.2011) (Region C – Cyclonic and Region D – Extreme cyclonic). Locations within these wind regions are likely to have the relevant environmental processes to cause extreme storm surge and coastal inundation associated with cyclonic events. While cyclone affected areas are prone to the largest storm tide events, coastal inundation from non-cyclonic low pressure systems (or other weather events) may also be relevant in some areas. This may be evidenced by pre-existing hazard mapping or historical events of coastal inundation due to abnormally high ocean water levels.

Step 2: Are there assets at risk?

The hazards that are considered in this section are natural events. In areas without significant human development (largely 'natural'), these flooding events do not necessarily have adverse consequences for human infrastructure and can be important to natural processes and ecosystem health. For instance, catchment flooding can bring nutrient-rich material that improves floodplain soils. As the physical accounts for flood, storm and erosion protection are to be measured in the number of properties protected through the restoration of a coastal wetland ecosystem, the second step of this assessment is to determine whether there is human infrastructure or assets which are at risk of damage from the hazard prior to the restoration activities.

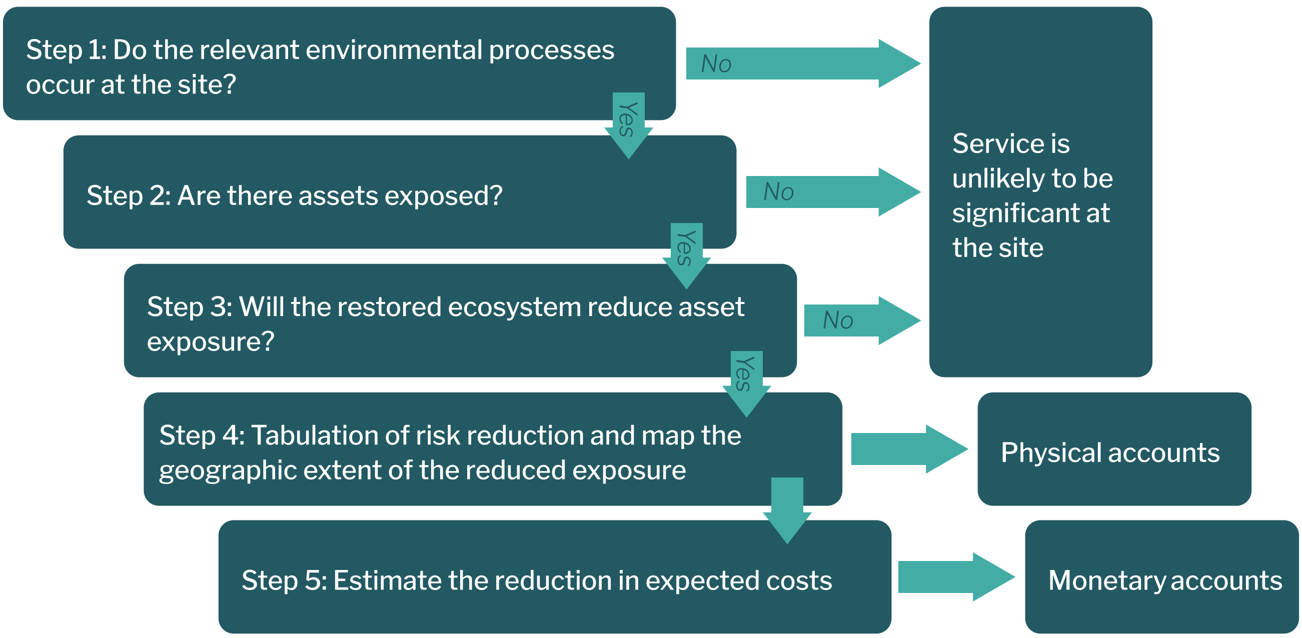

Figure 9.3: Flow chart for accounting for erosion, storm and flood mitigation services.

Persistent erosion

Understanding assets at risk from persistent erosion is based on an assessment of the land that is eroding. Where the land is developed, privately owned or socially/culturally valuable, erosion can be considered as causing a loss of an asset. However, where the land has limited human development and natural movement of banks or shorelines is considered acceptable and not resulting in a loss of valued land or cultural site, this indicates that there are no assets at risk.

Riverine flooding

Riverine flooding can occur on a catchment scale. As a result, flood processes occurring in one section of a floodplain (e.g. where a coastal restoration project is occurring) may influence flood levels elsewhere in the catchment. Therefore, the assets or built infrastructure at risk may be anywhere in the connected floodplain system. Typically, Australian riverine systems with significant infrastructure at risk will have a pre-existing flood study, which maps the extent of flood hazards and typically estimates the level of risk to assets[147]. Where there is infrastructure impacted by flooding according to a flood study, or historical evidence of flood damages throughout the floodplain, there are assets at risk due to riverine flooding.

Coastal inundation associated with storm tides, particularly cyclonic events

Coastal inundation associated with extreme ocean events typically occurs immediately adjacent to the coast or coastal waterways. Coastal wetlands are only expected to provide protection from coastal inundation at locations where they act as a barrier directly between coastal processes and built infrastructure. Based on this understanding, there must be assets located behind the restored site (e.g. the restored site is within the flow path between the ocean and the assets) and these assets must be within the coastal inundation hazard zone. This will typically be evidenced by existing storm tide hazard mapping or historical events.

Step 3: Will the restored ecosystem reduce the assets at risk?

The purpose of step 3 is to assess the capacity of the restored site to alter the physical processes that caused the hazard (e.g. reducing flood levels or limiting erosion). Step three considers a set of mechanisms that are likely to reduce the risks of each of the three hazard categories. Other mechanisms for risk reduction that may not yet have been foreseen in this guide would also be acceptable, as long as they can be incorporated into process-based models to demonstrate their effect. Should modelling be required for alternative purposes (e.g. if flood modelling is required to ensure that the restoration project has no adverse impacts on flooding), then using those model results to demonstrate any change in hazards would be acceptable to indicate risk reduction with restoration of coastal wetlands.

Persistent erosion

Coastal ecosystems can assist in managing persistent erosion by reducing or reflecting the energy of the flows or waves before (or as) it reaches the land. This has the potential to occur where the coastal ecosystem is in the direct pathway of the physical process causing erosion. The ecosystem will typically be located directly on the banks or dunes (where root systems will provide some protection from erosion) or immediately in front of the eroding bank/dune, such that the ecosystem dissipates wave or flow energy through increased drag or friction prior to reaching the foreshore. Protection from persistent erosion can be relevant to restoration of all blue carbon ecosystems. Protection from persistent erosion may also occur due to restoration activities, such as fencing, where livestock access has been a key contributor to erosion processes.

Riverine flooding

There are two main mechanisms through which coastal wetlands can reduce flood processes on a coastal floodplain system:

[147] Geosciences Australia. Australian Flood Risk Information Portal (AFRIP) https://www.community-safety.ga.gov.au/data-and-products/afrip Accessed 10th April, 2023

1. **Increasing flood storage.**

   Flood storage areas are those outside of main floodway channels that are preferentially flooded (either intentionally or due to natural topographic features) to slow water flows and reduced impacts on assets in the main floodway. Flood storage is most likely to be altered through restoration activities that involve changes to major levees which historically prevented connectivity during floods. To assess the capacity of the restored area to increase flood storage, the pre-restoration elevation of any levees altered by restoration should be compared to design flood levels in adjacent flood channels. In events where the levees previously contained floodwaters, but will no longer in the restored configuration, changes to flood storage are possible to estimate. Increases in flood storage are most likely to be effective during minor to moderate flood events, when existing levee systems were sufficient to provide protection. In most extreme events, if existing levee systems were already overtopped, no additional flood storage is likely to be provided.

2. **Improving conveyance of flood waters.**

   Improving the conveyance of local flood waters away from the floodplain can provide more efficient pathways for the drainage of floodwaters from direct upstream catchment flooding. Improving conveyance is most likely to occur by restoration activities where there is construction of additional drainage networks or increased size of drainage infrastructure (e.g. additional or larger culverts or floodgates). If such works are not included in restoration activities, improved flood conveyance is unlikely to occur. While changes to channel and floodplain friction (e.g. removal of weeds) can improve conveyance, due to the naturally flat nature of coastal wetlands this is considered unlikely to have a significant impact on flood conveyance, except where key flood paths are cleared, or new flood paths are created.

While a site-specific assessment of potential flood impacts should be completed, restoration projects most likely to provide flood protection may have:

- Included project activities that altered levees that were previously used for flood mitigation; and/or

- Included project activities that increased the conveyance capacity of flow restrictions on major flood paths (e.g. culverts).

Where levee removal or enhanced conveyance capacity has been proposed for restoration, projects may have had to include flood modelling as due diligence to ensure there are no adverse impacts on flooding of the landscapes. Where such modelling has been completed, this may be used for building these accounts.

Coastal inundation associated with storm tides, particularly cyclonic events

Coastal wetland vegetation, in particularly mangroves and saltmarsh, attenuate extreme water levels during cyclonic storm surge events. This primarily occurs due to the increased drag force or frictional forces leading to head losses as water flows through the wetland. Like riverine flooding, coastal inundation can also be influenced by project activities that alter flood storage, such as the modification of natural or artificial levees.

Development of a conceptual understanding of inundation pathways of the areas surrounding the restoration site will assist in assessing whether the restored system provides attenuation of extreme water levels. Restored coastal wetlands can attenuate coastal inundation through increased frictional forces in situations there are assets at risk landward of the restored coastal ecosystem. Greater hydraulic attenuation rates are achieved where storm durations are relatively short (less than 24 hours, which is likely in tide-dominated storm tides), where ecosystems are relatively dense, uniform and close to the surface, and where the relevant flow direction is not through well defined channels[148]. For example, while seagrass ecosystems may result in some minor attenuation of flow velocity, the position of these ecosystems within the water column (below lowest astronomical tides) results in less effective protection during extreme storm surge events[149] and this service is more likely to be provided by mangroves, saltmarsh or supratidal ecosystem restoration.

Any length of saltmarsh, mangroves or supratidal forest in the flow path of storm surge has the potential to attenuate water levels upstream. However, the amount of attenuation is dependent on the extent and characteristics (e.g. density) of the restored ecosystem, as well characteristics of the storm tide (e.g. storm duration). From a practical standpoint, it is likely that proponents will make a judgement on the potential scale of attenuation based on attenuation rates in scientific literature to assess the need to progress to a study quantifying the protection provided. While the body of literature estimating attenuation of storm tides is growing rapidly, studies have suggested that attenuation of storm surge inundation from mangroves is typically up to 50 cm per kilometre of ecosystem in the flow path[150;151;152] and from saltmarsh up to 25 cm per kilometre[153;154]. Where

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[148] Temmerman, S., et al. (2023). Marshes and mangroves as nature-based coastal storm buffers. Annual Review of Marine Science, 15: 95-118. https://doi.org/10.1146/annurev-marine-040422-092951

[149] Temmerman, S., et al. (2023). Marshes and mangroves as nature-based coastal storm buffers. Annual Review of Marine Science, 15: 95-118. https://doi.org/10.1146/annurev-marine-040422-092951

[150] Temmerman, S., et al. (2023). Marshes and mangroves as nature-based coastal storm buffers. Annual Review of Marine Science, 15: 95-118. https://doi.org/10.1146/annurev-marine-040422-092951

[151] McIvor, A. L., et al. (2012). Storm surge reduction by mangroves. Natural Coastal Protection Series: Report 2. Cambridge Coastal Research Unit Working Paper 35.

[152] Zhang, K., et al. (2012). The role of mangroves in attenuating storm surges. Estuarine, Coastal and Shelf Science 102:11-23. https://doi.org/10.1016/j.ecss.2012.02.021

[153] Leonardi, N., et al. (2018). Dynamic interactions between coastal storms and salt marshes: A review. Geomorphology, 301: 92-107. https://doi.org/10.1016/j.geomorph.2017.11.001

[154] Temmerman, S., et al. (2023). Marshes and mangroves as nature-based coastal storm buffers. Annual Review of Marine Science, 15: 95-118. https://doi.org/10.1146/annurev-marine-040422-092951

the lateral length of the restored ecosystem is small (in the order of 10 to 100 m, in the primary direction of the inundation flow), the impacts may be considered negligible, unless there are highly sensitive assets at risk behind the ecosystems and even small increases in protection are valuable. Consideration should be given to whether small scale changes (in the order of a few centimetres of storm surge) result in practical reduction in risks to assets prior to proceeding to a modelling approach to quantify risk reduction for small-scale restoration projects.

Like riverine flooding, coastal inundation may also be influenced by changes to or construction of engineering structures (e.g. levees) that change inundation patterns during extreme storm surge through the provision of additional flood storage. Where restoration activities have resulted in changes in flood storage in areas impacted by storm surge (described in the riverine flood section), this would also qualify as a mechanism to reduce coastal inundation, and further consideration of the quantification of benefits may be justified.

Step 4: Physical quantification of reduced asset exposure

There are two broad types of hazards dealt with in this section: hazards associated with long-term, ongoing processes (persistent erosion) and hazards associated with extreme events (riverine flooding and coastal inundation). Long-term, persistent processes are typically measured, following a consistent and reliable approaches. Such monitoring may require specialty equipment or knowledge, however it is expected this can be assessed at relatively low costs.

When assessing the impacts hazard reduction associated with extreme events, it is necessary to incorporate the restoration site into process-based modelling approaches. Process-based models are mathematical approximations of physical processes and can vary from simplistic empirical models to detailed numerical modelling. The impact of coastal ecosystems on physical processes is site specific, not readily generalised and cannot be easily monitored due to the infrequency and variability of extreme events. This is consistent with the recommendations of Beck et al. (2016)[155], as well as the Australian Rainfall and Runoff Guidelines[156] for assessing the effectiveness of flood mitigation measures. Modelling of the reduction of hazards associated with restoration requires the engagement of specialists in the field, capable of running and interpreting the required models. While the following sections provides recommendations for how risk reduction should be incorporated into the accounts, this guidance is not a prescriptive process limited to a single model formulation or type. It is anticipated that professional judgement from suitably qualified individuals will be used to ensure models are fit for purpose.

Persistent erosion - measurement

Persistent erosion is characterised by medium to long (greater than 12 months) term recession of banks or dunes due to sediment mobilisation driven by regular high water velocities or waves (including wind or boat wake waves). The quantification of persistent erosion requires baseline monitoring, measuring the rate of erosion prior to restoration works. Monitoring should map the position of the eroding bank/dune scarp through time.

Examples of acceptable methods for monitoring erosion include:

- Use of GPS monitoring equipment to map bank or dune scarp.
- High resolution aerial imagery (including drones or commercial products such as NearMaps, providing it can be adequately geo-rectified). Other sources of aerial imagery may be available on a case-by-case basis.

[155] World Bank. (2016). Managing Coasts with Natural Solutions: Guidelines for Measuring and Valuing the Coastal Protection Services of Mangroves and Coral Reefs. M. W. Beck and G-M. Lange (Eds.). Wealth Accounting and the Valuation of Ecosystem Services Partnership (WAVES), World Bank, Washington, DC.

[156] Ball, J. E., et al. (2016). Australian Rainfall and Runoff-A guide to flood estimation. Commonwealth of Australia.

- Satellite derived shorelines, such as those provided in the Digital Earth Australia (DEA) Coastlines[157], may be appropriate where the rate of change is sufficient to be quantified in satellite products. Where satellite derived products are used, multiple years of data should be assessed as erosion on a short time scale is unlikely to be quantifiable at the resolutions available.

Where aerial imagery is used to estimate erosion, care must be taken to account for differences in tide levels on shallow slopes. The area of bank scarp eroded between measurements should be quantified, and normalised in m^2 per year. Baseline monitoring must include at least 3 observations over a minimum of a 12-month period providing evidence of a persistent erosion trend. Note that longer periods of measurement are encouraged, as long as there are at least three observations in the last 12 months, and at least 1 observation annually for any prior period. The accuracy of the measurement technique needs to be commensurate with the scale of the erosion observed and will likely require sub 0.5 m accuracy in the horizontal direction.

Following implementation of the restoration project, continued observations of (previously) eroding landscapes should continue, ideally using the same method utilised for baseline monitoring (e.g. GPS monitoring, aerial imagery), with at least three measurements over a 12-month period. Similar to the baseline data, the area of bank scarp should be quantified, and normalised in m^2 per year. The difference between the two rates (measured in m^2 per year) can be assessed as the area of avoided erosion in the annual accounts. As coastal wetlands can also be accretionary environments (e.g. the foreshore accretes beyond its pre-restoration location), this may also include area of land accreted, where monitoring can show evidence of land accretion. Land accretion may also be included in the physical accounts as an area of m^2/year where it is observed.

Riverine flooding – numerical modelling

Industry best practice for estimating flood risk is through development of a numerical model that estimates flood depths across a floodplain. Throughout Australia, developed floodplains typically have pre-existing flood studies, which are vital for development planning and hazard mapping. Flood models in Australia are built in accordance with the Australian Rainfall and Runoff Guidelines[158](ARR) often held by local government authorities. Flood studies involve the development of models capable of simulating hydrologic and hydraulic behaviour within catchments to estimate flood behaviour (including flood levels and flows) across a specified domain. While the models are generally calibrated to observed events, it is standard practise in planning to also model 'design events'. Design flood events represent a statistical estimate of flooding based on an analysis of available driving environmental data (e.g. rainfall). Design flood events are assigned an Annual Exceedance Probability (AEP) which is the likelihood of a certain flood event (or a worse event) happening in a given year. It is anticipated that coastal ecosystems will have the most potential to provide flood mitigation benefits in minor to moderate floods (10 % and lower AEP).

Under the ARR guidelines[159], changes to flood risks from development (e.g. infrastructure) should be assessed by incorporating the development into existing calibrated and validated flood models and assessing changes in flood behaviour for a range of design events (e.g. AEP). A similar approach is to be used in assessing potential benefits due to site level restoration activities. Where existing flood models exist, the model geometry should be updated to reflect the restoration activities, in particularly any changes to potential flood mitigation infrastructure (e.g. levees or culverts). Wetland vegetation is incorporated into flood models through roughness parameters, which can be considered in the post restoration scenario. The use of models will require the support of specialist modellers capable of modifying and running existing flood models.

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[157] Geoscience Australia (2023) DEA Coastlines https://cmi.ga.gov.au/data-products/dea/581/dea-coastlines

[158] Ball, J. E., et al. (2016). Australian Rainfall and Runoff-A guide to flood estimation. Commonwealth of Australia.

[159] Ball, J. E., et al. (2016). Australian Rainfall and Runoff-A guide to flood estimation. Commonwealth of Australia.

Flood models should be re-run for event horizons where the restoration activities were likely to have influenced flood processes (most likely 10 % and lower AEP). Relevant event sizes should be assessed on a case-by-case basis, and should consider:

- Which events the project area was already drowned in the pre-restoration flood mapping. In these cases, restoration is unlikely to increase flood storage, and changes to flood mitigation may not be relevant.
- Existing flood levels compared to the elevation of infrastructure (e.g. levees) which were modified as a result of the restoration works.

For most locations where flood processes are relevant for accounts (i.e. there are mechanisms for the restored ecosystem to change flood processes), standard due diligence procedures may require flood modelling to assess potential increases in flood risks to neighbour properties or infrastructure. Where modelling is required, careful choice of model scenarios will allow the results of model runs to inform quantification of the physical accounts.

For riverine flooding, the physical accounts reflect the number of properties provided with additional protection, compared to the pre-restoration configuration. This is depicted conceptually in Figure 9.1. For each applicable flood event horizon, resultant flood depths across the model domain should be compared by subtracting the pre-restoration modelled flood depth from the post-restoration flood depth and mapped. Based on this mapping, the number of protected properties (e.g. buildings) should be counted using either aerial imagery, lot boundaries layers or land use layers. The physical accounts should be measured as the number of properties provided with reduced flood impacts (maximum value across all event horizons considered).

In the case of riverine flooding, infrastructure that supports the coastal wetland restoration (e.g. changes to levees, floodgates, or other flood mitigation infrastructure) may influence flood processes, not just the presence of the vegetation. Such changes should be considered in this assessment and be accounted for as 'Unattributed to specific ecosystem' where the restoration activities can be specifically identified as the driver for the change in risk to infrastructure.

Coastal inundation associated with storm tides, particularly cyclonic events

Storm surge during cyclonic events occurs due to a mixture of barometric set up, wind and wave setup, wave runup and astronomical tides. Like flood modelling, best practice for understanding coastal inundation from storm tides in Australia is through the use of process-based models[160], capable of predicting both[161]:

- An assessment of extreme coastal water levels. This includes estimating offshore hydrodynamics and characterising nearshore transformations.
- Inundation modelling and mapping, characterising the anticipated flood depths associated with the extreme coastal water level event.

While these two aspects of coastal inundation from extreme storm surge are inter-related, in practice, these two processes should be accounted for separately. To assess reduced asset exposure from coastal inundation due to coastal wetland restoration, which are located onshore or nearshore, inundation modelling and mapping cover the relevant processes for most restoration projects. As such, it is appropriate for existing boundary conditions (e.g. outputs from of extreme coastal water level assessment) to be used as a boundary condition for an inundation model with and without the restored coastal ecosystem where this is available.

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[160] DES. (2018). A guide to 'good practice' storm tide inundation mapping and modelling. Department of Environment and Science. Queensland Government, Brisbane.

[161] DEHP. (2013). Coastal hazard technical guide: Determining coastal hazard areas. Department of Environment and Heritage Protection. Queensland Government, Brisbane.

Best practice for inundation mapping uses process-based modelling to simulate flow paths, wave propagation, overland flow and coastal inundation. The numerical models used are similar to flood models, and the same model may be used if an appropriate model domain is selected. Modelling will require the use of specialist engineers with an understanding of both coastal processes and numerical modelling. Where available, existing inundation models (flood models) could be utilised and modified to assess changes in coastal inundation levels for a range of event horizons resulting from inundation from the sea (rather than runoff from the land). Models used in this application should be capable of incorporating the impact of attenuation from vegetation (e.g. mangroves or saltmarsh) typically through roughness or drag parameters (e.g. Manning's n). The models must be capable of predicting inundation behaviour both before restoration, and after restoration, including both differing roughness parameters, and structural changes (e.g. changing to levees or floodplain connectivity) related to restoration works. Like riverine flood modelling, models are re-run for multiple event horizons (e.g. AEP) where suitable boundary conditions exist. Physical accounts should be measured as the maximum number of properties provided with reduced coastal inundation levels based on a comparison of pre- and post- restoration inundation mapping (shown diagrammatically in Figure 9.2).

While coastal inundation from storm tides is well recognised as a hazard in cyclone affected areas in Australia, modelling and mapping of extreme storm tides is far less comprehensive than riverine flooding, mainly due to limited resources and monitoring data[162]. Some of the simplified methods commonly used to estimate the extent of coastal inundation from such events are not appropriate for quantification of changed risk from site level coastal wetland restoration projects. For example, many locations in Queensland use a static "bathtub method" to map storm surge risks, based on an elevation threshold of 1.5 m or 2.0 m above the highest astronomical tide[163]. Bathtub methods do not include the complexities of attenuation of water levels due to topographic or roughness constraints, including coastal wetland vegetation and assume that the water surface will be horizontal like a bathtub. Using these bathtub methods is not suitable for assessing the reduction in extreme storm surge associated with coastal wetland restoration.

[162] DES. (2018). A guide to 'good practice' storm tide inundation mapping and modelling. Department of Environment and Science. Queensland Government, Brisbane.

[163] DEHP. (2013). Coastal hazard technical guide: Determining coastal hazard areas. Department of Environment and Heritage Protection. Queensland Government, Brisbane.

Step 5: Monetary quantification of reduced asset exposure

The physical accounts for coastal protection services address the number of properties (or area of land, in the case of persistent erosion) that are provided with additional protection from inundation or erosion processes due to the presence of a restored wetland. A readily accepted way to estimate the value of this risk reduction is through the calculation of avoided damages (using an Expected Damage Function)[164]. Avoided damage cost is one of the methods suitable for estimating exchange values recognised in the SEEA-EA (para 9.52[165]). Avoided damages from extreme events can be calculated through changes in Annual Average Damages (AAD). AAD is a concept commonly used in flood management and involves the calculation of an equivalent annual expense if hazard damages occurred evenly throughout time. Calculation of AAD requires overlaying mapping produced in the physical quantification of the benefits with spatially explicit flood damage curves. This is aligned with the methods described in the Australian Rainfall and Runoff Guidelines[166], which requires flood risk mitigation effectiveness to be quantified through changes in the AAD, and the recommendations by the World Bank in their guidance for estimating coastal protection benefits of mangroves and reefs[167]. Similar methods should be applied, where possible, when assessing both flood and cyclone related benefits. Evaluation of persistent erosion services, however, may follow a more simplistic approach where avoided land losses are valued based on land valuations without the requirement to consider probabilistic events.

Persistent erosion

As persistent erosion is a long-term process, event probability does not need to be considered. Monetary accounts can be estimated as the value of the area avoided erosion multiplied by the unimproved capital value ($/m^2) as stated by the relevant state valuer general for the location in question. Where a lot/parcel specific land value cannot be determined, average land value within the local government area may be used.

In specific cases where restoration of coastal wetlands was completed instead of structural intervention (e.g. rock armouring of riverbanks), a replacement value (construction cost of rock-armouring) may be used in lieu of avoided damages. There must be evidence that structural interventions were considered to ensure there was demand for the protection delivered.

Riverine flooding and coastal inundation associated with storm tides in cyclone affected areas

As both riverine flooding and coastal inundation are both forms of flood events, the process for monetary quantification is similar, and thus they have been combined in this section.

Flood or inundation damages to properties should consider, at a minimum, structural damages, and damages to contents. These damages associated with a particular event horizons (e.g. AEP) are estimated using stage-damage functions, which uses idealised relationships that relate the depth of flooding above floor levels to the expected damage (see Figure 9.4). Some local government areas or locations may have developed site specific stage-damage curves for existing flood studies. Where such curves exist, they should be used. An example of recently developed tools for estimating flood damage is outlined in the NSW Flood Risk Management Guide (and associated excel documentation, currently in draft)[168]. This

[164] Menendez, P., et al. (2018). Valuing the protection services of mangroves at national scale: The Philippines. Ecosystem services, 34:24-36. https://doi.org/10.1016/j.ecoser.2018.09.005

[165] United Nations, et al. (2021). System of Environmental-Economic Accounting— Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

[166] Ball, J. E., et al. (2016). Australian Rainfall and Runoff-A guide to flood estimation. Commonwealth of Australia.

[167] World Bank. (2016). Managing Coasts with Natural Solutions: Guidelines for Measuring and Valuing the Coastal Protection Services of Mangroves and Coral Reefs. M. W. Beck and G-M. Lange (Eds.). Wealth Accounting and the Valuation of Ecosystem Services Partnership (WAVES), World Bank, Washington, DC.

[168] DEP (2022). Flood Risk Management Measures: Flood Risk Management Guide MM01. Department of Planning and Environment, New South Wales Government.

tool provides stage-damage curves (including structural, contents and intangible damages) for residential and commercial buildings, with suitable adjustments encouraged where there are regionally appropriate values available (such as average building size, average building replacement cost (per m^2)). For simplicity, the monetary quantification can focus solely on properties which have had a reduction in flood risk, rather than at a catchment wide scale (as would be required for a typical flood study).

Stage-damage curves allow for damage costs to be estimated per event for both flooding and coastal inundation. However, these are events that do not happen frequently. In some years, there may be no damage costs (due to no, or limited flood events), while in other years the extreme floods can cause catastrophic damages. To balance the complexities of the timing of the events, the ARR Guidelines[170] recommend the use of Average Annual Damages (AAD) to quantify flood damages to communities, as well as the effectiveness of flood risk mitigation actions. AAD is a concept used in flood management and involves the calculation of an equivalent annual expense if hazard damages occurred evenly throughout time. AAD accounts for both probability of an event and scale of damages. It is recommended that the monetary quantification of properties protected from riverine flooding or coastal inundation is estimated through the reduction in AAD (i.e. the avoided damage).

Methods for calculating AAD are described in detail in the ARR Guidelines[171]. however it broadly requires:

1. For each event horizon considered, use the stage-damage functions to estimate the avoided damages associated with the restoration works.
2. Plot the change in expected damages (y-axis) against the AEP (x-axis).
3. The area under this curve can be calculated to estimate the avoided AAD, measured in dollars per year.

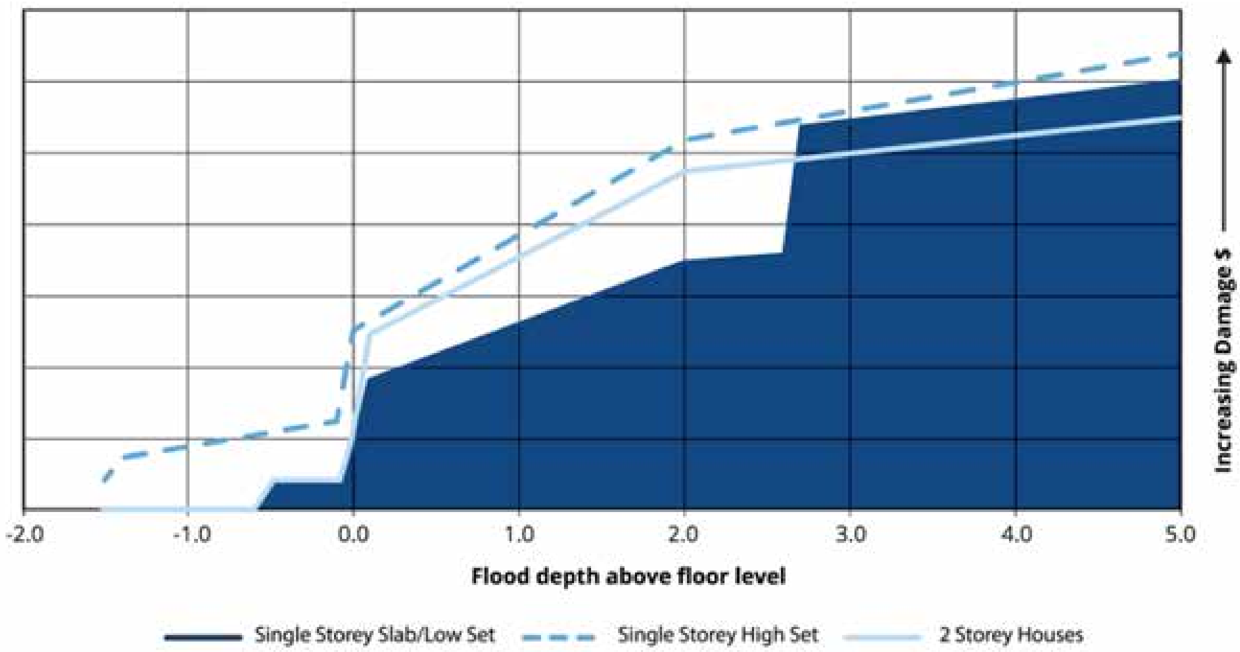

Figure 9.4: Flow chart for accounting for erosion, storm and flood mitigation services.

[169] Ball, J. E., et al. (2016). Australian Rainfall and Runoff-A guide to flood estimation. Commonwealth of Australia.

[170] Ball, J. E., et al. (2016). Australian Rainfall and Runoff-A guide to flood estimation. Commonwealth of Australia.

[171] Ball, J. E., et al. (2016). Australian Rainfall and Runoff-A guide to flood estimation. Commonwealth of Australia.

Note that this assessment must include events for which there is no expected change in expected damages (event if they are not explicitly modelled), which can be assessed qualitatively based on the capacity of the restored ecosystem to alter flood processes.

Data sources

The data sources required (and available) will vary depending on which of the coastal protection services are being considered for a project. Specific data, references and information required are discussed throughout the methods section for these services. However, where moderation of extreme events is to be quantified, it is recommended that users of this guide assess the presence of existing process-based models for flood or coastal inundation. Such models are managed by local government authorities (usually developed by external consultants) and are used for planning purposes. Use of existing models will significantly reduce the cost associated with process-based modelling.

Uncertainties

Models designed for coastal inundation or riverine flooding should adhere to national guidelines, such as the Australian Rainfall and Runoff guidelines. It is essential to engage experienced practitioners to ensure model reliability and confidence. Standard flood modelling practice typically requires model calibration using observed events to evaluate its ability to replicate critical processes accurately. Although some degree of uncertainty is inherent in all modelling, assessing relative change by comparing model scenarios, as recommended by the methods above, remains an effective approach when a model is 'fit-for-purpose'. Key assumptions, like friction or drag parameters for coastal wetland ecosystems, should undergo sensitivity tests to ensure robustness in model predictions.

The accuracy of data collection methods for measuring erosion, whether from riverbanks or dunes, will directly influence the level of uncertainty in estimating erosion protection services. The precision of methods used for tracking bank sediment movement (e.g. GPS measurements, imagery, or satellite-derived data) must match the scale of erosion being observed. For example, satellite-derived products with low to moderate resolution, typically in the order of tens of meters, cannot accurately quantify erosion on a 1-meter scale. To minimise uncertainty in quantifying erosion trends, it is essential to employ suitable datasets and provide evidence of consistent erosion patterns. This approach helps ensure more reliable assessments of erosion trends.

Key assumptions or limitations

Methods for measuring and modelling of extreme events associated with these coastal protection services from blue carbon ecosystems are still emerging. Process based models are well positioned to provide accurate, reliable estimates of the physical risk reduction associated with these services, however they require specialist skills and software, and can be costly to run. As such, it is likely benefits may only be quantified in situations where there is sufficient cause to assume they will have a significant impact on risk reduction (and therefore a high monetary value). Minor benefits may occur but may be either too costly or too uncertain to quantify with the existing methods.

While there is increasing literature showing the capability of coastal wetlands to mitigation extreme events, the methods used to estimate their impacts on a local, site scale project are still evolving. Similarly, the computing capacity and ability of numerical models to represent their impact is also developing. As such, this section in the guide itself will remain generic, and allow flexibility depending on the needs of the project (e.g. to do complex numerical modelling) based on location, importance to stakeholders, and project size. Coastal wetlands may be used in lieu of, or in conjunction with conventional structural flood, erosion, or storm defence in which case such detailed modelling may be required for other aspects of the project and should be used.

10. Fish production: nursery habitat service

Dr Daniel E. Hewitt, Prof. Rod M. Connolly, Dr Matthew D. Taylor, Dr Vincent Raoult, Dr Paul Carnell

10.1 Summary of section

Coastal wetlands have a well-established role as being the basis for harvestable recreational and commercial fisheries species and for coastal marine animals generally. They perform this function through two main pathways 1) as habitat in the juvenile (nursery) and adult phases and 2) as primary productivity as the basis for the coastal food web (food). Here, this section focuses on the nursery capacity of restored wetlands and their role in increasing numbers of fish and invertebrates. This is an intermediate service that does not directly link to a user/beneficiary and provide an exchange value, however, because it is the foundation of fishery-related benefits, it allows a more complete interpretation of the benefits of restoring wetlands to commercial and recreational fisheries at the project level. For example, a restored habitat may not provide large amounts of biomass provisioning services, while providing a significant nursery habitat service. To fully assess the benefit of blue carbon ecosystem restoration, it is therefore helpful to develop accounts for nursery habitat services as well as biomass provisioning.

While the role of coastal wetlands in acting as nursery grounds is well established, the methods for integrating nursery benefits into accounts are still being developed; there is general agreement on basic principles[172], but methods continue to be refined and improved[173,174,175]. This need for further refinement stems from two issues: 1) While

[172] Blandon, A. & zu Ermgassen, P. S. (2014). Quantitative estimate of commercial fish enhancement by seagrass habitat in southern Australia. Estuarine, Coastal and Shelf Science. https://doi.org/10.1016/j.ecss.2014.01.009.

[173] zu Ermgassen, P. S., et al. (2016). Quantifying fish and mobile invertebrate production from a threatened nursery habitat. Journal of Applied Ecology. https://doi.org/10.1111/1365-2664.12576.

[174] Jänes, H., et al. (2020). Quantifying fisheries enhancement from coastal vegetated ecosystems. Ecosystem Services. https://doi.org/10.1016/j.ecoser.2020.101105.

[175] Carnell, P., et al. (2022). Prioritising the restoration of marine and coastal ecosystems using ecosystem accounting. Preprint: ResearchSquare. https://doi.org/10.21203/rs.3.rs-1617940/v1.

previous approaches have estimated the increased number of individuals or biomass of fish and invertebrates, this only represents a commercial or recreational ecosystem service once the fish is caught; 2) For harvestable species, the guide proposes measures of increased juvenile animal abundances in restored habitats, with subsequent modelling of contributions to the biomass of harvestable stocks as an index of enhancement value (e.g. kilograms per hectare per year).

The method presented here, formulated in Peterson et al.[176] and extended in Blandon and zu Ermgassen[172] and zu Ermgassen et al.[173], represents the combination of a series of calculations based on well-understood biological processes. Measured juvenile densities (individuals per unit area) are used to estimate the density of fish in subsequent year classes (e.g. 1 years old, 2 years old, etc.) based on estimated mortality rates. The average length of a fish in each year class is then estimated using the von Bertalanffy growth function and converted to biomass based on known length-weight relationships. Multiplying this biomass by the density of fish in each year class gives an estimate of the total biomass per unit area in that year class. Finally, this is multiplied by the area of each ecosystem to give an estimate of the increased nursery service (i.e. fish production) in each year surveyed, so that when repeated over time will resulting from habitat restoration.

It is possible to achieve estimates of this sort using two approaches; a higher cost approach based on collecting field data and lower cost approach using existing data. Firstly, proponents may wish to plan and execute their own field program sampling juvenile fish in wetlands, with an estimated cost of ~ AUD$100 k for field sampling. Secondly, proponents may wish to draw on existing datasets from similar wetlands or meta-analysis (if sufficient data is available)[172,174], with no associated field costs. The primary differences between these two approaches relate to data collection (e.g. sampling design), the inferences that can be drawn from a given dataset and the time to completion (typically the latter will be much faster). While resource availability is likely to be a primary driver of the choice between these approaches, careful consideration should be given to the suitability of existing data, and consultation with relevant experts (e.g. local fisheries scientists, statisticians) is encouraged. The use of existing data will be most effective where that includes a time series showing trends in the use of relevant habitats after restoration work, so that a temporal component can be retained in creating accounts. Importantly, the analytic framework remains the same for each and is described in the following, along with some general guidelines for planning and conducting fieldwork.

10.2 Detailed section

Methods

What to collect

In general, the nursery habitat concept is defined in relation to harvested species[177], however the guide also encourages recording the nursery service for non-harvested species which may contribute to other benefits as an intermediate service (i.e. Fish important for dive tourism). This may vary between restoration projects and consultation with local fisheries scientists can inform which species are likely to be present.

How to collect it

The sampling design should capture the natural variability in juvenile fish densities in wetlands. For example, recruitment to coastal wetlands can vary temporally and spatially, sometimes in relation to environmental factors (e.g. wind-driven patterns in circulation[178]). As a result, sampling at different times and/or places can result in vastly different estimates of juvenile density. To counter this, sampling should be carried out many times at

[176] Peterson, C. H., et al. (2003). Estimated enhancement of fish production resulting from restoring oyster reef habitat: quantitative valuation. Marine Ecology Progress Series. https://doi.org/10.3354/meps264249.

[177] Beck, M. W., et al. (2001). The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. BioScience. https://doi.org/10.1641/0006-3568(2001)051[0633:Ticamo]2.0.Co;2.

[178] Taylor M. D., et al. (2017). Recruitment and connectivity influence the role of seagrass as a penaeid nursery habitat in a wave dominated estuary. Science of the Total Environment. https://doi.org/10.1016/j.scitotenv.2017.01.087.

both the restoration site and control (or reference) sites[179], both pre- and post-restoration. This allows the effects of restoration to be isolated from natural variability, and capture periods of high and low recruitment, thereby defining a range of plausible outcomes from habitat restoration (i.e. 'best' and 'worst' case scenarios). This type of study design is commonly referred to as a before-after-control-impact (or BACI) design (see Underwood[180] and references therein). Optimal sample sizes and sampling frequencies are related to the expected variability in a system, and consultation with local fisheries scientists and statisticians is necessary when designing a field program to ensure sampling effort is adequate to capture expected changes.

A critical aspect of experimental design is the choice of sampling gear[181]. Two key considerations apply to gear selection, it should: (1) be 'spatially explicit', sampling individuals from a known area (e.g. ha, m^2) specific to a certain habitat type (e.g. mangroves); and (2) have a known catch efficiency (i.e. probability of capturing an individual), thereby facilitating comparison between different gear types used in different habitats. See Rozas and Minello[181] and Harrison-Day et al.[182] for discussion of commonly used gear types in estuarine systems.

Estimating biomass

Most studies estimating the nursery value of blue carbon ecosystems enhancement, which is the extra fish produced within an ecosystem (e.g. Seagrass) compared to a sandy area, or its degraded state. However, using the SEEA-EA accounting framework, requires this to be done in a way that the biomass production for each ecosystem type is calculated at each accounting time point (e.g. Before restoration, 5 years after restoration). In this instance, the density for each species s, is converted to biomass production (*BE*; kg per unit area) by first estimating the density of juveniles surviving to age class i (D_i) using the following equation (to simplify notation we have not included subscripts that denote species/sex, but separate calculations should be made where species- and sex-specific parameters are available):

$$D_i = D_{0.5}e^{-M_{(i-0.5)}}$$

where $D_{0.5}$ is the juvenile density and M is natural mortality. If no published estimates of natural mortality are available, it can be estimated using the approach in Then et al.[183]:

$$M = 4.899t_{max}^{-0.916}$$

where t_{max} is the maximum age for a species. Following this, the length of an average fish in each age class *i* (L_i; cm) can be calculated using the von Bertalanffy growth equation:

$$L_i = L_\infty\,(1 - e^{(-K(i-t_0))})$$

where L_∞ is the asymptotic maximum average length, *K* is the Brody growth coefficient and t_0 is the theoretical age when length is 0. Lengths are then converted to weights (W_i; g) based on:

$$W_{i,s} = a_s L_{i,s}{}^{b_s}$$

where *a* and *b* are the intercept and slope, respectively, of the length-weight relationship. Biomass in each year class *i* (*B*; kg) is then calculated using the following equation:

$$B_i = W_i D_i$$

From this, the incremental increase in biomass between years classes (ΔB_i; kg) is calculated:

$$\Delta B_i = B_i - B_{i-1},$$
$$\Delta B_{i=t_{harvest}} = B_{i=t_{harvest}}$$

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[179] Boys, C. A. & Pease, B. (2016). Opening the floodgates to the recovery of nektonic assemblages in a temperate coastal wetland. Marine and Freshwater Research. https://doi.org/10.1071/MF15445.

[180] Underwood, A. J. (1992). Beyond BACI: the detection of environmental impacts on populations in the real, but variable, world. Journal of Experimental Marine Biology and Ecology. https://doi.org/10.1016/0022-0981(92)90094-Q.

[181] Rozas, L. P. & Minello, T. J. (1997). Estimating densities of small fishes and decapod crustaceans in shallow estuarine habitats: A review of sampling design with focus on gear selection. Estuaries. https://doi.org/10.2307/1352731.

[182] Harrison-Day, V., et al. (2021). A systematic review of methods used to study fish in saltmarsh flats. Marine and Freshwater Research. https://doi.org/10.1071/MF20069.

[183] Then, A. Y., et al. (2014). Evaluating the predictive performance of empirical estimators of natural mortality rate using information on over 200 fish species. ICES Journal of Marine Science. https://doi.org/10.1093/icesjms/fsu136.

where $t_{harvest}$ is the age of recruitment to the fishery. From this, total harvestable biomass (*HB*; kg ha^{-1} y^{-1}) can be calculated by summing the incremental increase in weight between year classes from the age of first harvest to the maximum age of the species (t_{max}):

$$HB = \sum_{i=t_{harvest}}^{t_{max}} \Delta B_i$$

Finally, enhancement of fish production due to restoration (E; kg y^{-1}) can be calculated:

$$E = HB\ \Delta Area$$

Where Δ*Area* is the change in habitat area (ha) due to restoration.

Using model outputs to account for nursery service

Given that the nursery service is measured and then modelled as biomass growth into the future, this data needs to be handled appropriately for input into ecosystem accounts. We suggest that while models can be run based on data from one accounting period, as demonstrated above, the biomass outputs can only be incorporated into accounts by aligning the age of the ecosystem (particularly relevant for restoration) and the age class. For example, while sampling in seagrass five years after restoration might support an estimated potential biomass production of 115 kg per hectare, per year, this is across all age classes of fish. Thus, only the age classes up to 5 years old should be included in the estimates of fish biomass.

Nursery service as an Intermediate service

Given that the measurement of blue carbon ecosystems as a nursery for juvenile fisheries species does not include a direct link to known users/ beneficiaries, it is classified as an Intermediate service. Below in Table 10.1 we show an example whereby, the supply of the intermediate service of nursery habitat is attributed to seagrass and there is a use of the nursery service by mangrove and saltmarsh ecosystems (as an input to its supply of final ecosystem services) and supply of fisheries biomass provisioning services (as outlined below in Section 11). There is no double counting implied through the recording of intermediate services since the user of the intermediate service is different from the user of the associated final ecosystem service.

Table 10.1: **Ecosystem services physical supply and use table, demonstrating the integration of nursery services as an Intermediary service to biomass provisioning.** IS= Intermediate service. Grey cells indicate not applicable

		Economic units			Ecosystem assets		
Supply	**Units of measure**	**Fishers**	**Gov.**	**House-holds**	**Mangroves**	**Saltmarsh**	**Seagrass**
Fisheries biomass provisioning service	Tonnes				400	600	200
IS: Nursery service	Tonnes						1000
Use							
Fisheries biomass provisioning service	Tonnes	1200					
IS: Nursery service	Tonnes				400	600	

Data sources

Whether undertaking primary data collection, or using previously collected data, the general information required to assess the provision of fish nursery habitats includes:

1. Habitat extent pre- and post-restoration (e.g. ha, m^2).
2. Densities of juvenile fish species (individuals per unit area) pre- and post-restoration.
3. Life-history parameters for fish species sampled (see **Estimating**).

Habitat extent can be measured following the procedures described in **Section 5: Ecosystem extent**. If undertaking primary data collection, densities of juvenile fish species are the main data to be collected. Otherwise, juvenile densities can be taken from long-term monitoring in similar systems (as demonstrated in our case-study) or from a meta-analysis[174]. Such data may be available in the primary literature, however in some cases it may held by the relevant fisheries agency. Similarly, life-history parameters can be sourced from the primary literature, grey literature (e.g. fisheries agency reports) and public repositories although the suitability of parameters needs to be carefully considered, as outlined below). A crucial step when planning data collection and curation is the engagement of relevant experts, such as local fisheries scientists, who can provide guidance regarding sampling design, and the applicability of data sources and types.

Key assumptions or limitations

Two central assumptions in this analysis are (1) that fisheries in the location where habitat restoration has taken place are limited by availability of suitable habitat, rather than by larval supply or other factors dampening recruitment; and (2) that productivity increases linearly with increases in habitat. These assumptions underpin the representation of fisheries enhancement via nursery services as a constant value per unit area[173]. In systems where much of the habitat has been lost this assumption is likely to hold, either in enclosed bays and estuaries or more open coastal waters. However, where the restored area is very substantial, at some undefined point habitat will no longer be the limiting factor and this approach would no longer be appropriate[173]. The precise nature of the relationship between habitat extent and fisheries productivity is currently unclear and is likely to vary on a location-by-location basis. Also note that if fish biomass projections are subsequently used to inform economic accounts, the area from which fishery catch data are obtained must be relevant for the juvenile fish at the restoration location. Consultation with fisheries experts with local knowledge during project planning is vital to assess the suitability of this assumption and thus this approach.

Species-specific population processes are also likely to influence the response of exploitable populations to habitat restoration. For example, freshwater inflow influences spawning, recruitment, and growth in several estuarine species[184,185] and can also influence the distribution of harvests (e.g. estuarine or coastal)[186]. Processes such as density-dependent predator-prey dynamics will also influence the realized benefits of increased biomass provisioning through habitat restoration.

Finally, it is important to ensure the life-history parameters used are representative of the population being considered. For example, zu Ermgassen et al.[173] showed that the approach can be sensitive to parameter choice (e.g. different mortality rates), in some cases leading to biomass

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[184] Hewitt, D. E., et al. 2022. Crabs go with the flow: Declining conductivity and cooler temperatures trigger spawning migrations for female Giant Mud Crabs (*Scylla serrata*) in subtropical estuaries. Estuaries and Coasts. https://doi.org/10.1007/s12237-022-01061-1.

[185] Ruello, N. (1977). Migration and stock studies on the Australian school prawn Metapenaeus macleayi. Marine Biology. https://doi.org/10.1007/BF00394025.

[186] Gillson, J., et al. 2012. Effects of flood and drought events on multi-species, multi-method estuarine and coastal fisheries in eastern Australia. Fisheries Management and Ecology. https://doi.org/10.1111/j.1365-2400.2011.00816.x.

enhancement estimates varying by a factor of 2. Therefore, data from public data repositories[187] should be used with caution; while they can be useful, in some instances the parameters presented correspond to a specific region and may not generalize well.

Uncertainties

Calculations of nursery habitat service are expected to be applied to species for which reliable data on juvenile densities are available. Whether the data are collected on site as part of restoration monitoring or drawn from relevant literature values, the mean estimates will also have measures of variation (such as confidence limits or standard deviations). These estimates of variability can and should be included throughout the series of equations used to model fish biomass, so that the level of uncertainty is quantified. Biological parameters necessary to complete the modelled estimates, such as natural mortality and age at maturity, can vary among locations. Where this type of variability is known it also can be included in the biomass estimates. Collectively, the propagation of these different sources of variation through the modelling steps offers a clear method for quantifying uncertainty in biomass estimates.

[187] https://www.fishbase.se/search.php

11. Fish production: biomass provisioning service

Dr Daniel E. Hewitt, Prof. Rod M. Connolly, Dr Matthew D. Taylor, Dr Vincent Raoult, Dr Paul Carnell

11.1 Summary of section

Restoration of coastal ecosystems create and enhance flows that support commercial fisheries. Establishing direct and indirect links between restored coastal ecosystems and commercial fisheries provides the foundation for both physical and monetary accounts. Restored aquatic habitats result in the creation of complex structural habitats within the intertidal zone, and adjacent subtidal areas. As noted above, these habitats provide additional high-quality 'space' that may be directly occupied by juvenile and adult life stages of exploited species. However, as the aquatic plant community establishes and matures within restored habitats, this indirectly benefits ecosystem services for exploited species through two key additional functions across the broader estuarine ecosystem. Firstly, these plants support regulatory and maintenance functions that improve water quality (detailed above in Section 10), which leads to improvements in the health and productivity of estuarine faunal communities. Secondly, and the component focused on here, the additional primary production created by these plants enhances the estuarine food web in which fisheries species feed, and thus these fisheries species are reliant on for their diet and growth. This is classed as indirect, because often fisheries species may not directly consume coastal wetland plants, but they feed on animals that do, which makes them indirectly reliant on coastal wetland plants.

Indirect links from this trophic subsidy to fisheries productivity are likely to capture most of the flows of ecosystem services arising through restored or enhanced blue carbon ecosystems. The flow of provisioning services to exploited species can be estimated through trophic modelling. This provides a basis for the physical accounts - exploitable fish biomass derived from primary productivity synthesised within a particular restored habitat. This approach allows the proportion of commercially exploitable fish biomass (kg; from catch statistic datasets) derived from a particular ecosystem asset to be estimated on a species-by-species basis, and expressed as an area unit (e.g. kilograms per hectare per year). This can then be extrapolated using the areal coverage of the improved ecosystem asset and integrated over the relevant time frame. The physical accounts can be translated into monetary accounts - the value of this exploitable fish biomass at first-point-of-sale (FPS) which can incorporate estimation of other flow-on economic outputs that follow the capture of seafood (e.g. processing, supply chain, retail). This occurs using market-based and other economic data and may be forecast based on expected future productivity and changes in market values.

The method presented here, originally developed in Taylor et al.[188], represents the integration of a set of straightforward relationships linking the trophic provisioning services of primary producers within coastal wetlands (e.g. mangroves, saltmarsh) with fisheries harvest. Stable isotope analysis is used to estimate the proportional contribution of plant species to the diet of commercially harvested species. These proportions are then multiplied by the annual harvest to apportion it among these plants/ecosystems. Harvested biomass is then multiplied by the landing value of these species to give an estimate of the economic contribution these habitats make on a per unit area basis. Finally, this estimate is multiplied by the areal increase in habitat resulting from restoration to estimate the economic value of habitat restoration to commercial fisheries. The method has been most commonly applied in the estuarine habitats of NSW but stable isotope data are widely available for many other coastal ecosystems in Australia[189], including coastal seagrass meadows[190], and these can inform restoration accounting so long as the harvested biomass is known for a defined area relevant to restoration works.

At the time of writing, studies using this approach have focused on estimating the value of coastal wetlands to the commercial fishing sector[170]. This is because economic and harvest data are readily available for commercially exploited species. In principle, the same analysis could be applied to the recreational sector if reliable economic and harvest data is available[191].

It is possible to achieve estimates of this sort using two approaches. Firstly, proponents may wish to plan and conduct a field program for sampling commercially harvested species in coastal wetlands (including stable isotope analysis) with estimated field costs of ~$AUD 100k. Secondly, proponents may draw on existing stable isotope datasets from similar systems, with no associated field costs (demonstrated in the Hunter case study[192]). The primary differences between these two approaches relates to data collection (e.g. sampling design), the inferences that can be drawn from a given dataset, and the time to completion (typically the latter will be much faster). While resource availability is likely to be a primary driver of the choice between these approaches, careful consideration should be given to the suitability of existing data, and consultation with relevant experts (e.g. local fisheries scientists, statisticians, stable isotope ecologists) is encouraged. Importantly, the analytic framework remains the same for each and is described in the following, along with some general guidelines for planning and conducting fieldwork.

[188] Taylor, M. D., et al. (2018). The economic value of fisheries harvest supported by saltmarsh and mangrove productivity in two Australian estuaries. Ecological Indicators. https://doi.org/10.1016/j.ecolind.2017.08.044.

[189] Abrantes KG, Barnett A, Baker R, Sheaves M (2015) Habitat-specific food webs and trophic interactions supporting coastal-dependent fishery species: an Australian case study. Rev Fish Biol Fish 25:337-363

[190] Connolly RM, Hindell JS, Gorman JD (2005) Seagrass and epiphytic algae support nutrition of a fisheries species, Sillago schomburgkii, in adjacent intertidal habitats. Mar Ecol Prog Ser 286:69-79

[191] Taylor, M. et al. In review. Valuing provision services of coastal wetlands for recreational fisheries. Ecological Indicators.

[192] Glamore, W., et al. (2023). Accounting for benefits from coastal restoration: a case study from the Hunter River. Report to DCCEEW.

11.2 Detailed section

Methods

What to collect

Proponents should seek to include exploited species (or 'consumers') that comprise as great a proportion of the overall harvest from a given estuary as possible, which can be informed directly via catch data or consultation with representatives of local fishing co-operatives[193]. Samples should be adult individuals of a commercially harvestable size, which can generally be obtained from commercial operators within an estuary or local fishing co-operatives[193]. In the absence of these options, fishery-independent sampling is required, and the precise methods will vary depending on target species. Any species harvested well beyond the estuary should not be considered, as these require a more complex model of the habitat–fishery linkage[194].

Primary producers (or 'sources') sampled should include all plant sources potentially providing basal energy and nutrients to food webs supporting the exploited species. The suite of primary producers will vary between restoration projects, however Australian studies typically include samples of seagrass, saltmarsh grasses and succulents, mangrove leaves, epiphytic algae (e.g. on seagrass, mangrove pneumatophores), particulate organic matter (POM), and fine benthic organic matter (FBOM)[195]. POM can be collected by towing a 200 µm plankton net, while FBOM samples are obtained by collecting the top centimetre of sediment into sample jars for further processing. All samples should be stored on ice for transportation to the laboratory, and frozen until further processing.

Optimal sample sizes and sampling frequencies are related to the expected variability in a system, model complexity (e.g. individual-level variation in consumers)[196] and desired precision of estimates[197,198]. Simulations can be useful to assess optimal sample size[199] but may be beyond the scope of some projects. At a minimum, statisticians and stable isotope ecologists should be consulted to ensure sampling effort is adequate to reliably estimate source contributions.

How to process samples

Muscle tissue is the most commonly sampled tissue, owing to a relatively long isotopic turnover in fish[200] and crustaceans[201] making it a good indicator of long-term assimilated diet. For fish, samples can be taken from dorsal muscle, while crustacean

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[193] Jänes, H., et al. (2022). The value of estuarine producers to fisheries: A case study of Richmond River Estuary. Ambio. https://doi.org/10.1007/s13280-021-01600-3.

[194] Camp, E. V., et al. (2020). Impacts of habitat repair on a spatially complex fishery. Estuarine, Coastal and Shelf Science. https://doi.org/10.1016/j.ecss.2019.02.007.

[195] Hewitt, D. E., et al. (2020). Stable isotopes reveal the importance of saltmarsh-derived nutrition for two exploited penaeid prawn species in a seagrass dominated system. Estuarine, Coastal and Shelf Science. https://doi.org/10.1016/j.ecss.2020.106622.

[196] Semmens, B. X., et al. (2009). Quantifying inter- and intra-population niche variability using hierarchical Bayesian stable isotope mixing models. PLOS ONE. https://doi.org/10.1371/journal.pone.0006187.

[197] Phillips, D. L., et al. (2014). Best practices for use of stable isotope mixing models in food-web studies. Canadian Journal of Zoology. https://doi.org/10.1139/cjz-2014-0127.

[198] Brett, M. T. (2014). Resource polygon geometry predicts Bayesian stable isotope mixing model bias. Marine Ecology Progress Series. https://doi.org10.3354/meps11017.

[199] Pearson, J. & Grove, M. (2013). Counting sheep: sample size and statistical inference in stable isotope analysis and palaeodietary reconstruction. World Archaeology. https://doi.org/10.1080/00438243.2013.820646.

[200] Suring, E. & Wing, S. R. (2009). Isotopic turnover rate and fractionation in multiple tissues of red rock lobster (Jasus edwardsii) and blue cod (Parapercis colias): consequences for ecological studies. Journal of Experimental Marine Biology and Ecology. https://doi.org/10.1016/j.jembe.2008.11.014.

[201] Hewitt, D. E., et al. (2021). Diet-tissue discrimination and turnover of δ13C and δ15N in muscle tissue of a penaeid prawn. Rapid Communications in Mass Spectrometry. https://doi.org/10.1002/rcm.9167.

muscle tissue is most easily sampled from chelipeds for crabs, or the abdomen for prawns. Care should be taken to isolate the chosen tissue and may involve manual separation (e.g. organs, carapace fragments)[202], chemical extraction (e.g. acid-washing)[203,204] or mathematical correction of stable isotope ratios[205]. Once processed samples should be dried before being ground into a fine powder and placed into aluminium caps for stable isotope analysis (1–2 mg for animal tissue and 6–8 mg for plant tissue).

Stable isotope analysis

Stable isotope analysis (also referred to as 'isotope ratio mass spectrometry') is used to determine the ratio of heavy to light isotopes of an element in a sample (Rsample). This ratio is then compared to universal standards (Rstandard) and reported using δ-notation (in parts-per-mil or ‰):

$$\delta X = [(R_{sample} / R_{standard} - 1)] \times 10^3$$

where R is given by:

$$R = F_{heavy} / F_{light}$$

and F is the fractional abundance of the isotopes of element X.

An important consideration when planning a study is which stable isotopes (i.e. 'tracers') to use. Carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) in 'bulk' tissue are by far the most common[206] and should be included at a minimum. However, there are many options available which can enhance an analysis. For example, sulphur ($\delta^{34}S$) can be useful for differentiating between seagrass and saltmarsh contributions[207,195], which generally have similar $\delta^{13}C$ signatures. Compound-specific stable isotope analysis (e.g. amino acids) is also a useful avenue for differentiating among plant types not easily separated using 'bulk' isotope ratios[208]. In general, cost is the limiting factor when it comes to the inclusion of additional isotopes; $\delta^{13}C$ and $\delta^{15}N$ are the cheapest at ~ $AUD 20 per sample, while $\delta^{34}S$ costs ~$AUD 50 per sample, and ~ $AUD 100 per sample for compound-specific stable isotope analysis.

Stable isotope mixing models

Basic stable isotope mixing models assume that, for a given isotope, the isotopic signature of a consumer (δ_c) is given by:

$$\delta_c = \sum_{i=1}^{s} f_i(\delta_i + \gamma_i)$$

where f_i is the proportional contribution of the ith source, δ_i is the isotopic signature of the ith source and γ_i is the isotope-specific fractionation of the ith source[209]. The most recent development of these models has been within a Bayesian statistical framework, allowing flexible model specification that can incorporate prior knowledge of animal diet (e.g. from gut content analysis)[209], uncertainties in source and consumer stable isotope signatures,

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[202] Mazumder, D., et al. (2008). Variability of stable isotope ratios of Glassfish (Ambassis jacksoniensis) from mangrove/saltmarsh environments in southeast Australia and implications for choosing sample size. Environmental Bioindicators. https://doi.org/10.1080/15555270802266003.

[203] Raoult, V., et al. (2019). Resource use of great hammerhead sharks (Sphyrna mokarran) off eastern Australia. Journal of Fish Biology. https://doi.org/10.1111/jfb.14160.

[204] Schlacher, T. A. & Connolly, R. M. (2014). Effects of acid treatment on carbon and nitrogen stable isotope ratios in ecological samples: a review and synthesis. Methods in Ecology and Evolution. https://doi.org/10.1111/2041-210x.12183.

[205] Post, D. M., et al. (2007). Getting to the fat of the matter: models, methods and assumptions for dealing with lipids in stable isotope analyses. Oecologia. https://doi.org/10.1007/s00442-006-0630-x.

[206] Fry, B. (2006). Stable isotope ecology, Springer. https://doi.org/10.1007/0-387-33745-8.

[207] Connolly, R. M., et al. (2004). Sulfur stable isotopes separate producers in marine food-web analysis. Oecologia. https://doi.org/10.1007/s00442-003-1415-0.

[208] Ramirez, M. D., et al. (2021). Meta-analysis of primary producer amino acid δ15N values and their influence on trophic position estimation. Methods in Ecology and Evolution. https://doi.org/10.1111/2041-210X.13678.

[209] Moore, J. W. & Semmens, B. X. (2008). Incorporating uncertainty and prior information into stable isotope mixing models. Ecology Letters. https://doi.org/10.1111/j.1461-0248.2008.01163.x.

concentration dependence and many possible sources[197]. Common examples of these models include 'MixSIR'[209], 'SIAR'[210] and 'MixSIAR'[211] all of which are implemented within the R statistical computing environment[212].

Phillips et al.[197] present an extremely thorough best-practice review that is essential reading during the planning phase of a stable isotope study. Briefly, it is important to: (1) include all possible sources; (2) ensure all consumer stable isotope signatures are within the 'mixing polygon' (i.e. the region in δ-space bounded by the source isotope signatures)[213]; (3) ensure the number of sources is less than the number of tracers + 1[210], which may necessitate grouping of sources; and (4) correct for trophic fractionation (i.e. the differential accumulation of particular isotopes in consumer tissues) in the mixing model[197].

Estimating biomass provisioning

Biomass provisioning ($B_{h,s}$; kg y^{-1}) from habitat h to species s is estimated using the following formula:

$$B_{h,s} = C_{h,s} H_s$$

where $C_{h,s}$ is the estimated proportional contribution of habitat *h* to species *s* (based on an isotope mixing model) and H_s is the annual harvest of species s (kg y^{-1}). This value represents the physical account for this service, which is then used to compile the monetary account in terms of the gross value of product ($GVP_{h,s}$; AUD y^{-1}) using the following formula:

$$GVP_{h,s} = B_{h,s} M_s P_s$$

where M_s is the consumer price index (CPI) corrected market value at first-point-of-sale (AUD kg^{-1}) and P_s is a fixed spatial partitioning coefficient for species *s*. The spatial partitioning coefficient is a subjective estimate of the average proportion of total harvest that is taken within the modelled region that can take any value between 0 and 1. This parameter is included to account for the relevant section of the estuary used by species s, effectively constraining estimates of $GVP_{h,s}$. For each species, P_s should be informed via expert opinion and consultation with fishers and fisheries compliance officers regarding the distribution of catch and effort within the system of interest.

From GVPh we estimated the ecosystem service value of habitat h (ESVh) by deducting direct operational costs (OC) of fishing, which for example has been estimated for New South Wales estuarine fishing (52 % of revenue) and reported in Voyer et al. [214]. These data are not always available for some fisheries, however, and since fishing efficiency increases with time, using recent, geographically appropriate values is important to represent the true value of the ecosystem service. This step is necessary to separate market value of product from the value of the product (and additional production) to the fishers themselves.

While the further economic contribution of the fishery to society is not included in the SEEA-EA tables, it is possible to estimate this to provide an indication of these additional benefits. Expected flow-on economic benefits of commercial fishing (e.g. retail and processing output) can be estimated by calculating the total economic output for habitat h and species s ($TEO_{h,s}$) using the following formula:

$$TEO_{h,s} = GVP_{h,s} m$$

[209] Moore, J. W. & Semmens, B. X. (2008). Incorporating uncertainty and prior information into stable isotope mixing models. Ecology Letters. https://doi.org/10.1111/j.1461-0248.2008.01163.x.

[210] Parnell, A. C., et al. (2010). Source partitioning using stable isotopes: coping with too much variation. PLOS ONE. https://doi.org/10.1371/journal.pone.0009672.

[211] Stock, B. C. & Semmens, B. X. (2016). MixSIAR GUI User Manual. 3.1 ed.

[212] R Development Core Team (2023). R: A language and environment for statistical computing. Vienna Austria: R Foundation for Statistical Computing.

[213] Smith, J. A., et al. (2013). To fit or not to fit: evaluating stable isotope mixing models using simulated mixing polygons. Methods in Ecology and Evolution. https://doi.org/10.1111/2041-210X.12048.

[214] Voyer, M., et al. (2016). Social and economic evaluation of NSW coastal professional wild-catch fisheries: valuing coastal fisheries, Fisheries Research and Development Corporation.

where m ($N[\mu, \sigma]$) represents an economic multiplier, derived from the relationship between statewide-GVP for the relevant state, and the minimum (TEO_{min}) and maximum (TEO_{max}) estimates of total economic output from commercial fishing, estimated according to:

$$\sigma = \frac{TEO_{max} - TEO_{min}}{6GVP}$$

$$\mu = \frac{TEO_{max}}{GVP} - 3\sigma$$

For each habitat *h*, biomass provisioning ($B_{h,s}$), gross value of product ($GVP_{h,s}$) and total economic output ($TEO_{h,s}$) are summed across all species s to give their cumulative value (B_h, GVP_h and TEO_h, respectively) which is then divided by the areal extent (ha) of habitat h within the model region to give habitat-specific estimates on a per unit area basis (e.g. kg ha^{-1} y^{-1} and AUD ha^{-1} y^{-1}). Finally, these values are multiplied by the area of each ecosystem *h* (ha) to obtain an estimate of the for each ecosystem type. When this is done before and after restoration, this will show the impact of ecosystem restoration on commercial fisheries biomass.

Both published examples of this method[170], conduct this analysis within a Monte Carlo simulation framework, whereby the model is fit *n* = 5,000 times with parameters randomly drawn from their respective distributions on each iteration. It is possible to conduct this analysis using point estimates (e.g. average annual harvest, etc.), however the Monte Carlo approach should be preferred as it preserves the full range of plausible outcomes from habitat restoration (i.e. 'best' and 'worst' case scenarios), while requiring little additional effort.

Data sources

A crucial step when planning data collection and curation is the engagement of relevant experts (e.g. local fisheries scientists, stable isotope ecologists) who can provide guidance regarding sampling design, and the applicability of data sources and types. Four types of data are needed to assess biomass provisioning from blue carbon ecosystem restoration sites:

1. Estimates of the proportional contribution of each habitat to the diet of exploited species. This data is either collected from the study site using the higher cost approach, or using existing values from other locations or meta-analysis[215].

2. Annual commercial harvest data for the estuary (e.g. kg y^{-1}). See Table 11.1.

3. Market price at first-point-of-sale (e.g. AUD kg^{-1}) for each species. See Table 11.1.

4. Ecosystem extent pre- and post-restoration (e.g. ha, m^2) in the restoration area, and in the broader estuary.

[215] Jänes, H., et al. (2020). "Stable isotopes infer the value of Australia's coastal vegetated ecosystems from fisheries." Fish and Fisheries, 21(1), 80-90. https://doi.org/10.1111/faf.12416

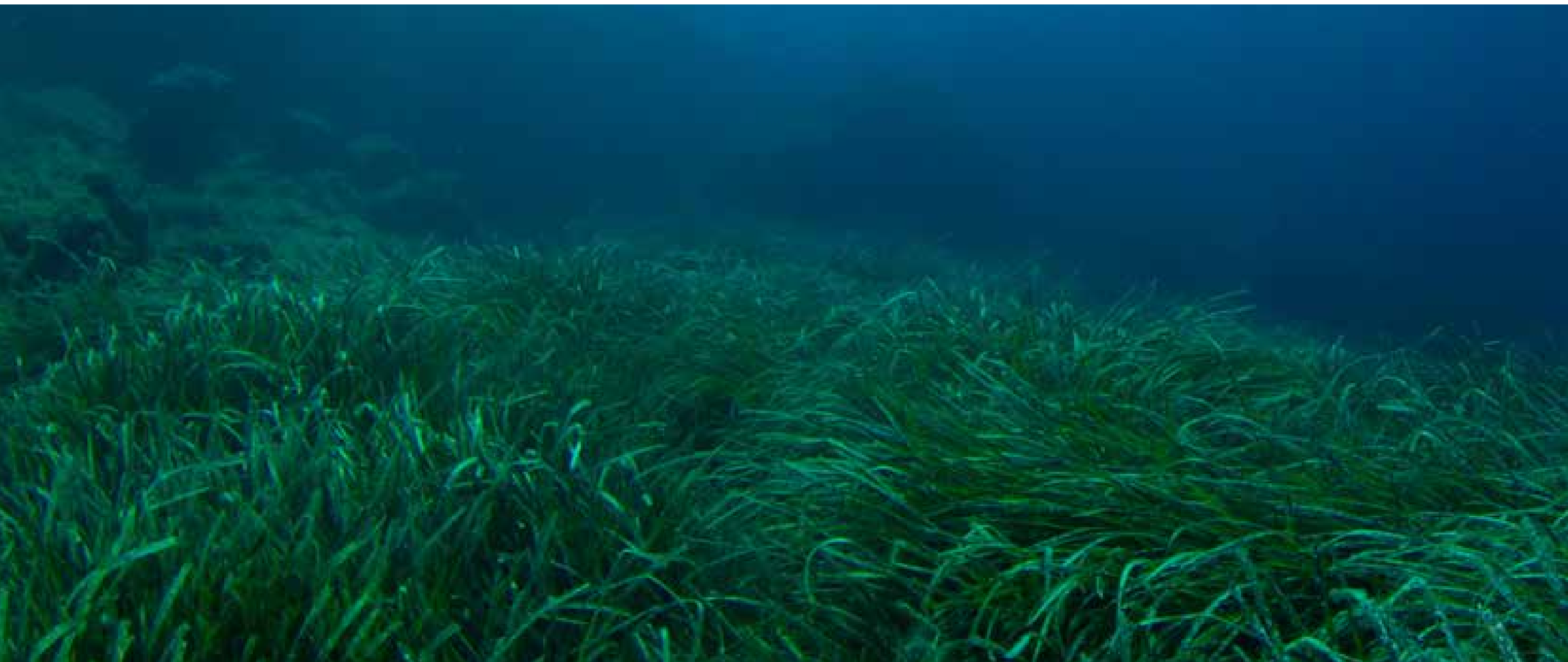

Table 11.1: Potential sources for data acquisition relating to commercial harvest and market price for exploited fish species.

Resource	Custodian	Jurisdiction/coverage	Link
Commercial harvest data	Department of Primary Industries Fisheries	New South Wales	https://www.dpi.nsw.gov.au/fishing/commercial/catch-effort
	Victorian Fisheries Authority	Victoria	https://vfa.vic.gov.au/commercial-fishing/commercial-fisheries
	Department of Natural Resources and Environment	Tasmania	https://fishing.tas.gov.au/commercial-fishing
	Department of Primary Industries and Regions	South Australia	https://www.pir.sa.gov.au/primary_industry/commercial_fishing
	Department of Primary Industries and Regional Development	Western Australia	https://www.fish.wa.gov.au/Fishing-and-Aquaculture/Commercial-Fishing/Pages/default.aspx
	Department of Primary Industries and Fisheries	Northern Territory	https://nt.gov.au/marine/commercial-fishing
	Department of Agriculture and Fisheries (QFISH)	Queensland	https://qfish.fisheries.qld.gov.au/
Market price	Sydney Fish Market	Australia	https://www.sydneyfishmarket.com.au/Seafood-Trading/Fish-Prices

Key assumptions or limitations

Two central assumptions in this analysis are (1) that fisheries within the estuary where habitat restoration has taken place are limited by availability of suitable habitat, rather than by larval supply or other factors dampening recruitment; and (2) that productivity increases linearly with increases in habitat. These assumptions underpin the representation of fisheries enhancement as a constant value per unit area[173]. In systems where much of the habitat has been lost this assumption is likely to hold. However, where the restored area is very substantial, at some undefined point habitat will no longer be the limiting factor and this approach would no longer be appropriate[173]. The precise nature of the relationship between habitat extent and fisheries productivity is currently unclear and is likely to vary on an estuary-by-estuary basis. Consultation with fisheries experts with local knowledge during project planning is vital to assess the suitability of this assumption and thus this approach.

Species-specific population processes are also likely to influence the response of exploitable populations to habitat restoration. For example, freshwater inflow influences spawning, recruitment, and growth in several estuarine species[184,185] and can also influence the distribution of harvests (e.g. estuarine or coastal)[186]. Processes such as density-dependent predator-prey dynamics will also influence the realized benefits of increased biomass provisioning through habitat restoration.

Finally, when employing the Monte Carlo approach, it is important to ensure that the distribution chosen for each parameter is appropriate. Typically, commercial harvest will follow a positive continuous distribution (e.g. log-normal or Poisson). Market price will most likely follow a normal (Gaussian) distribution, however if prices are generally low relative to the variance a normal distribution with truncation at 0 may be required. If fitting Bayesian mixing models, samples should be drawn from the posterior distributions of each source. However, if using previously published estimates positive continuous distributions truncated at 0 and 1 will be required. In cases where the proportional contributions are well estimated a truncated normal distribution may be appropriate, but sources with relatively low contributions may be better approximated by an exponential or Poisson distribution (see Hewitt et al.[216] for an example of this).

Uncertainties

While there are uncertainties associated with this approach, conservative steps are taken throughout biomass provisioning accounts, and Bayesian approaches have the benefit of being able to be clear about uncertainty levels. For example, total value of biomass is reduced to the 'fishable' portion of the biomass arising from restoration: this is a factor that greatly reduces the total value of habitats to conservatively apportion the fishable biomass. It is likely in practice that a larger portion of fishes produced by these habitats are in fact fishable. Using average values of fish product rather than year-on-year assessments means that market forces are unlikely to impact valuations. Any biomass accounts are therefore presenting conservative values.

The stable isotope approaches and the mixing models used to assess contribution of primary producers to the diets of commercially-important fishes are well established methods that have been validated in many species and taxa. In addition, these models include uncertainty calculations for each of the factors that may be included (for example, the trophic discrimination). These are automatically compounded and can be included in final valuations to provide a clear approximation of uncertainty.

[216] Hewitt, D. E., et al. (2020). Stable isotopes reveal the importance of saltmarsh-derived nutrition for two exploited penaeid prawn species in a seagrass dominated system. Estuarine, Coastal and Shelf Science. https://doi.org/10.1016/j.ecss.2020.106622

12. Cultural services: recreation and non-use values

Dr Abbie Rogers, A/Prof. Michael Burton, Dr Tafesse Estifanos, Dr Fitalew Taye

12.1 Summary of section

Cultural ecosystem services[217] include various non-material benefits people obtain from nature for recreational, spiritual, and psychological wellbeing[218]. Based on the Common International Classification of Ecosystem Services (CICES)[219], cultural services encompass characteristics of ecosystems that enable cultural benefits to be enjoyed. For example, coastal wetlands may provide cultural services such as nature-based recreation, aesthetic benefits, symbolic or spiritual benefits, as well as services that may not require use of natural assets such as the benefits derived from the knowledge that a specific natural ecosystem or wildlife exists (existence value). The SEEA framework in line with CICES defines recreation-related services as "the ecosystem contributions, in particular through the biophysical characteristics and qualities of ecosystems, that enable people to use and enjoy the environment through direct, in-situ, physical and experiential interactions with the environment. This includes services to both locals and non-locals (i.e. visitors, including tourists). Recreation-related services may also be supplied to those undertaking recreational fishing and hunting" (United Nations et al. 2021, p.133)[220].

[217] 'The term "cultural services" is not implied that culture itself is a service, rather it is a collective label intended to capture the variety of ways in which people connect to, and identify with, nature and the motivations attributed to these connection (United Nations et al., 2021 p.130).

[218] Millennium Ecosystem Assessment. (2005). Ecosystems and Human Well-being: Biodiversity Synthesis. W. World Resources Institute, DC.

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

[220] United Nations et al., (2021). System of Environmental-Economic Accounting—Ecosystem Accounting (SEEA-EA). White cover publication, pre-edited text subject to official editing. https://seea.un.org/ecosystem-accounting

The health and well-being, recreational and cultural ecosystem services associated with a specific site will be a function of the characteristics of the site, and how humans interact with it. SEEA-EA uses a range of economic valuation methods to measure these benefits[220,221]. This section outlines how recreational and non-use cultural services (e.g. existence values) may be monetised as exchange and welfare values, as well as other economic indicators (such as job creation).

12.2 Detailed section

Methods

Below, the guide first introduces the two value concepts – exchange and welfare values – that underpin EEA and economic valuation. Both focus on monetization of ecosystem services, enabling direct comparison of the value of biophysical assets with other economic assets. Economic valuation is a separate (but related) economic tool to the SEEA-EA and is discussed further in Appendix 2, but introduced here as it is the framework that underpins the use of welfare values. In the monetary accounts, SEEA-EA focusses on the use of exchange values, but for some services it is appropriate to report welfare values in bridging tables, particularly for non-use related values that cannot otherwise be reported directly in a monetary account, explained below.

The guide then discusses a range of non-market valuation methods that are relevant to estimate these values, acknowledging that there is complementarity between measuring welfare and exchange values is also discussed, before identifying additional economic indicators that can be worthwhile reporting in conjunction with EEA measures.

Exchange values for Environmental Economic Accounting

EEA is often applied at a national or regional scale to understand the contribution that environments make towards the productivity of a country. Being able to monitor whether these contributions are growing or declining over time allows us to understand environmental performance relative to economic performance, and whether society is succeeding in meeting sustainable development goals.

At a local level, EEA can assist with understanding the contribution of a smaller area, and/or specific environmental projects occurring in that area, to the productivity of a country or region. It can be used to clearly articulate in physical (and monetary) terms the impacts of a restoration project over time through use of repeated accounts tracking key indicators.

An important element of EEA is to provide biophysical measures of ecosystem services. The physical measures can be used to track the changes in stock of ecosystems used for services such as recreation, enabling the relative comparison of their performance with other economic stocks and flows.

The SEEA-EA uses the concept of 'exchange value' to do this. An exchange value represents the price at which goods or services can be exchanged. Where an asset is traded through a market, the exchange value is represented by its market price. Where it is not traded, an equivalent market value must be inferred.

The SEEA-EA also sets out a preference hierarchy of the valuation methods that should be used to estimate exchange values, consistent with how exchange values are calculated for the System of National Accounts as follows (SEEA-EA 9.23, p193; hereafter referred to as the "Exchange value hierarchy methods (i) to (v)"):

i. Directly observe the price for the ecosystem service if it is part of a market (e.g. price per ton of commercially caught fish). However, such a particular method may not be directly applicable in regulating and cultural services since there is no direct market for the services.

ii. Extrapolate prices that can be observed for similar ecosystem services that are part of a market (e.g. the price of fish sold in one coastal location is used to infer the price for the same species of fish in another coastal location where the fish are not being caught and sold).

[221] De Valck, J. & Rolfe, J. (2022). Reviewing the use of proxies to value coastal and marine biodiversity protection: The Great Barrier Reef in Australia. Marine Policy, 136, 104890. https://doi.org/https://doi.org/10.1016/j.marpol.2021.104890

iii. Use methods where the price for the ecosystem service is a component of an observable market transaction (e.g. use the hedonic pricing method to estimate the value of visual amenity for houses built on the coast).

iv. Use methods where the price for the ecosystem service can be estimated from revealed expenditures for related goods and services that are part of a market (e.g. use the travel cost method to estimate the recreational value of a wetland to visitors).

v. Use methods to estimate the price for the ecosystem service based on expected expenditures (e.g. the cost of replacing the ecosystem service with something else that would contribute the same type of benefits).

Welfare values for economic valuation

Welfare values reflect the contribution of a project to producer surplus and consumer surplus. Producer surplus is the profit generated through production of marketed goods (i.e. revenue less production expenses). It reflects that a producer will invest time and resources into production of a good or service because they are benefitting from that process. In principle, it is possible for a good to have a positive exchange value (i.e. market price) while generating zero (or even negative) producer surplus. Consumer surplus is a measure of the benefit to the consumer of a good or service, above and beyond the price paid for it (i.e. the difference between the maximum price they would have paid for each of the units of the good, and what they paid for it). It reflects that consumers are willing to give up money for a good because they gain more from their purchase than the value of the money spent. This can often mean that estimates of welfare value will be larger than estimates of exchange value for consumers of an ecosystem service.

The aggregation of the consumer and producer surplus components reflects the total social benefit of the project over its costs, or in other words – the 'total economic value' of a project. Welfare values are fundamental for economic valuation and associated assessment frameworks like benefit-cost analysis (see Appendix 2).

Welfare values are not formally a measure that can be balanced with other exchange values in a monetary account, since they go beyond the value of the exchange to the full value the service generates. However, they can be reported in bridging tables (see Section 12 of the UN SEEA-EA Guidelines for advice on how to prepare a bridging table). Noting that the same methodologies, with some nuances, often underpin estimation of both exchange and welfare values, where it is feasible to estimate welfare value as well, the inclusion of bridging tables may be useful for a number of reasons:

i. There may be services for which the exchange value is small and the welfare value is large, for example, a recreational site that is important for a local community, where the travel expenditures of visiting the site are small/negligible. The exchange value reported alone may not convey the importance of the site for its recreational services.

ii. Currently, there is no accepted method for estimating exchange values for non-use values, so if these are likely to be important for a restoration site then welfare values are the most appropriate representation of their monetary value.

iii. There are benefits in reporting welfare values in ways they can subsequently be referred to in other economic analysis valuations (e.g. benefit-cost analyses).

It may not be worthwhile estimating and including welfare values in SEEA-EA where:

1. The preferred method for estimating exchange value is not conducive to welfare valuation
2. Non-use values are not likely to be relevant to the site, for example, where site-stakeholders are unlikely to hold non-use values, or the project manager/investor is not interested in quantifying wider social benefits
3. Where there is no obvious need to prepare information that can assist in other economic valuations, social assessments of benefits and costs, or prioritisation of investments across projects.

Non-market economic valuation methods

Recognizing that many coastal wetland ecosystem services, particularly the regulating and cultural services, are not formally traded in a market, the calculation of both exchange and welfare values for such ecosystem services and assets depends on non-market valuation methods.

Non-market valuation[222,223,224]: includes economic methods that measure people's potential willingness to pay for environmental assets that are not typically bought and sold. Willingness to pay is a measure of the value an individual holds for an improvement (or to avoid a decline) in a social or environmental outcome, defined by how much of their disposable income they would be willing to trade for that outcome.

Non-market valuation approaches are designed to estimate welfare values (but adjustments or simulations can be used to derive values deemed equivalent to exchange values). The range of methods includes techniques that can measure all components of Total Economic Value, including both use and non-use values. Note that the preferred valuation approach for estimation of welfare values depends on which components of Total Economic Value (i.e. use and non-use values), are most relevant for the asset. The methods are elaborated here to guide their application for cultural services.

Non-market valuation methods are broadly categorized into stated and revealed preference approaches. Other alternative approaches include cost-based valuation and benefit transfer.

Revealed preference approaches: these are non-market valuation approaches that rely on observation of people's behaviour to understand the values associated with an environmental asset or site. The methods measure use-related values only. The most widely used methods include:

- **Travel cost method**[225,226]: the travel cost method is mostly used to value recreational services in outdoor recreation. The demand for a recreational site could be derived from the visitor's travel costs when visiting a specific site. Travel cost models based on primary studies are a well-established suite of models that can be employed to quantify the value of services that rely on visitation. Although these have been most widely applied for recreational activities such as recreational fishing[227], any form of visitation that generates value could be used as the basis for quantifying those values. The primary studies themselves can involve an increasing degree of complexity (zonal, individual and site choice models). The expenditure data collected in travel cost applications can often be used to directly estimate exchange values.

- **Hedonic pricing method**[228]: this is a method used to value environmental goods and services as components of the property value through analysing how the characteristics of an asset influence

[222] Champ, P. A. et al. (Eds.). (2003). A Primer on Nonmarket Valuation. Springer Science+Business Media. https://doi.org/10.1007/978-94-007-0826-6_2

[223] Rogers, A.A.& Burton, M.P. (2019). "Non-market valuation instruments for measuring community values affected by coastal hazards and their management". Report prepared for the Western Australian Department of Planning, Lands & Heritage by The University of Western Australia, Crawley.

[224] Baker, R. & Ruting, B. (2014). Environmental Policy Analysis: A Guide to Non-Market Valuation. In Productivity Commission Staff Working Paper.

[225] Willis, K. G. & Garrod, G. D. (1991). An Individual Travel-Cost Method of Evaluating Forest Recreation

[226] Czajkowski, M., et al. (2019). The Individual Travel Cost Method with Consumer-Specific Values of Travel Time Savings. Environmental and Resource Economics, 74(3), 961–984. https://doi.org/10.1007/s10640-019-00355-6

[227] Huang, B., et al. (2020). Quantifying welfare gains of coastal and estuarine ecosystem rehabilitation for recreational fisheries. Science of the Total Environment, 710, 134680. https://doi.org/10.1016/j.scitotenv.2019.134680

[228] Bishop, K. C., et al. (2020). Best practices for using hedonic property value models to measure willingness to pay for environmental quality. Review of Environmental Economics and Policy, 14(2), 260–281. https://doi.org/10.1093/reep/reaa001

its market value[229]. It is mostly applied to estimate the value of environmental goods based on the market price of a property. If the market price of a property with good environmental amenity is higher than the same type of property with respect to all attributes except for lower environmental amenity, the price difference could be attributed to the environmental amenity service.

Stated preference approaches: these are non-market valuation approaches that use survey-based methods to propose scenarios to people and ask them about their willingness to pay for preferred outcomes, or the trade-offs they are prepared to accept between different outcomes. Commonly used when non-use or 'existence' values are important, because it is possible to ask people about their values for environments even if they do not directly use them (i.e. even if behaviour cannot be observed).

- **Contingent valuation**[230,231]: A survey-based method in which values are elicited by directly asking respondents their willingness to pay for improvements (or avoid deterioration) in quality of specific environmental goods or services in a hypothetical market setting. For instance, asking respondents about their willingness to pay for restoration of a degraded wetland. The contingent valuation method can be undertaken in different formats including opened ended questions and dichotomous choices.
- **Discrete choice experiments**[232,233]: This is an attribute-based valuation technique in which the values for different features or attributes of an environmental good or services can be estimated based on choices made by respondents for different alternatives, often in hypothetical scenarios. Respondents are provided with a sequence of hypothetically designed choice cards with options consisting of combinations of identified attributes. For example, analysing preferences for different management scenarios of a coastal wetland with attributes such as increasing species diversity, improving water quality, improving recreational facilities, and cost of restoration. The willingness to pay for specific attributes of an environmental good or services could be estimated from the trade-offs made between gains (or avoided losses) of the environmental attributes and the price (cost) attribute included in the choice cards.

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229 Hanley, N. & Barbier, E.B. (2009). Pricing nature: cost-benefit analysis and environmental policy. Edward Elgar, Cheltenham

230 Boyle, K. J. (2017). Contingent Valuation in Practice BT - A Primer on Nonmarket Valuation (P. A. Champ, K. J. Boyle, & T. C. Brown (eds.); pp. 83–131). Springer Netherlands. https://doi.org/10.1007/978-94-007-7104-8_4

231 Johnston, R. J., et al. (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

232 Bennett J. and Balmey R. (2001). The Choice Modelling Approach to Environmental Valuation. Edward Elgar Publishing Ltd, Cheltenham, UK

233 Hoyos, D. (2010). The state of the art of environmental valuation with discrete choice experiments. Ecological Economics, 69(8), 1595–1603. https://doi.org/10.1016/j.ecolecon.2010.04.011

Valuation using secondary data (benefit transfer): A method to extrapolate willingness to pay estimates from primary studies that have used the above methods to secondary contexts where primary studies do not exist.

- **Benefit transfer**[234,235,236], which may be the most appropriate approach if resources limit the ability to conduct primary surveys, has an established methodological framework for identifying the best possible representation of value derived from existing literature. However, it can only be used in monetary accounts if the methods used have estimated exchange values. Benefit transfer methods can be broadly grouped into value transfers and function transfers. Unit value transfer is the process of estimating the value of an ecosystem service of interest (at the "policy site") by assigning a single value or a set of value estimates from an existing valuation study for a closely similar ecosystem and associated human population demographic elsewhere (at a "study site")[237,238].

  Unadjusted unit values (i.e. willingness to pay estimates directly taken from existing literature) are not seen as particularly reliable. Adjusted unit value transfers, where some changes are made to an estimate taken from a study site to match it better to the policy site (e.g. adjusting for average income of the relevant population demographic), can provide a good approximation of actual willingness to pay if the study and policy site decision contexts, environmental characteristics (including scale of the environmental changes occurring) and population demographics are very similar.

  Benefit function transfers identify factors that influence value reported in the literature and use a statistical function to predict an appropriate value for the site in question. This value function relates the value of an ecosystem service to the characteristics of the ecosystem and the beneficiaries of the ecosystem service and calculates the willingness to pay value of an ecosystem service at the policy site. The critical challenge to using benefit transfer is to account for the most important differences in the characteristics of the study and policy sites. Hence it is necessary to make adjustments to transferred values using important determinants of those estimated values.

Whichever of the methods above are used, whether a primary non-market valuation study or a benefit transfer, there has to be some evaluation of the size of the relevant population that is associated with the value (e.g. by identifying the level of visitation associated with a site for recreational services), and hence the basis for aggregation. If collecting primary data through a survey, it may be possible to also collect data to assist in identifying the relevant population. Alternatively, complementary sources of data can be consulted (e.g. visitation data collected by site managers).

Cost-based valuation: such methods help to estimate the value of ecosystem services based on avoided damage or replacement costs. These are not strictly welfare measures (they do not provide an estimate of consumer or producer surplus), and they are the least preferred (ranked 'v') of the exchange value hierarchy methods described above. However, where other information is not available, they may provide an indicative measure of value.

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[234] Wilson, M. A. & Hoehn, J. P. (2006). Valuing environmental goods and services using benefit transfer: The state-of-the art and science. Ecological Economics, 60(2), 335–342. https://doi.org/10.1016/j.ecolecon.2006.08.015

[235] Richardson, L., et al. (2015). The role of benefit transfer in ecosystem service valuation. Ecological Economics, 115, 51–58. https://doi.org/10.1016/j.ecolecon.2014.02.018

[236] Johnston, R.J., et al. eds (2015). Benefit Transfer of Environmental and Resource Values: A Guide for Researchers and Practitioners. Dordrecht, the Netherlands: Springer.

[237] Johnston, R. J., et al. (2021). Guidance to Enhance the Validity and Credibility of Environmental Benefit Transfers. Environmental and Resource Economics, 79(3), 575-624. https://doi.org/10.1007/s10640-021-00574-w

[238] Brander, L. (2013). Guidance manual on value transfer methods for ecosystem services. Nairobi, Kenya: United Nations Environment Programme (UNEP).

- **Avoided damage cost**[239]: This reflects the value ecosystem services provide in their ability to avoid or mitigate potential damage (and associated costs) that could have otherwise occurred. It is mostly used to estimate the value of ecosystem services protecting built capital assets from natural disasters, such as coastal storm protection or flood control services of mangroves and other vegetation.

- **Replacement cost**[240]: This uses the cost of replacing non-marketed ecosystem services as a proxy for the value of ecosystems. It is estimated based on the costs incurred in construction of man-made structures to replace the services provided by a specific ecosystem, for instance construction of floodwalls or levees for coastal storm protection. Man-made structures rarely replace the full set of services provided by natural ecosystems and hence values estimated based on replacement cost often does not imply the full value of the ecosystem services.

Finally, additional steps may be required to convert the welfare values derived from non-market valuation methods to exchange values.

- **Simulated exchange value method**: This approach estimates simulated exchange values from welfare values if site specific demand can be derived[241]. That means it is possible to estimate the exchange value (cost of the trip) of associated ecosystem services, such as for recreation-related services, based on demand function (estimated using the travel cost method or stated preference methods discussed above) and using existing data on actual number of trips to a recreational site.

  Presently, it is accepted that simulation can be used to derive exchange values from use-related welfare value measures, but the ability to simulate exchange values for non-use welfare values is debated (and hence why non-use values can only be reflected in bridging tables in SEEA-EA). The ways in which some stated preference studies are framed may preclude the conditions needed to reflect (and thus simulate) an exchange value; however, work is ongoing in environmental economics to reconcile exchange/welfare value concepts with respect to non-use values.

Some useful links to additional non-market valuation and benefit transfer resources are provided below.

- Baker and Ruting (2014). Environmental Policy Analysis: A Guide to Non-Market Valuation https://www.pc.gov.au/research/supporting/non-market-valuation

- Champ et al. (2003). A Primer on Nonmarket Valuation. https://link.springer.com/book/10.1007/978-94-007-0826-6

- Freeman et al. (2014). The Measurement of Environmental and Resource Values: Theory and Methods: https://doi.org/10.4324/9781315780917

- Rolfe et al., (2015) Introduction: Benefit Transfer of Environmental and Resource Values: https://link.springer.com/chapter/10.1007/978-94-017-9930-0_1

- Johnston, R.J., Rolfe, J., Rosenberger, R.S., and Brouwer, R., eds. Benefit Transfer of Environmental and Resource Values: A Guide for Researchers and Practitioners: https://link.springer.com/chapter/10.1007/978-94-017-9930-0_1

[239] Glass, C. A. & Burgess, D. E. (2023). Cost Benefit Analysis of a Catchment Management Scheme using the Avoided Cost Method. In Agricultural Economics Society, 97th Annual Conference 2023.

[240] Notaro, S. & Paletto, A. (2012). The economic valuation of natural hazards in mountain forests: An approach based on the replacement cost method. Journal of Forest Economics, 18(4), 318–328. https://doi.org/10.1016/j.jfe.2012.06.002

[241] Caparrós, A., et al. (2017). Simulated exchange values and ecosystem accounting: Theory and application to free access recreation. Ecological Economics, 139, 140-149. https://doi.org/10.1016/j.ecolecon.2017.04.011.

Complementarity in exchange and welfare values

While exchange and welfare values are different theoretical concepts, in practice the calculation of each type of value often relies on the same data and methods for data collection. For example, where ecosystem services are part of a formal market, the prices for services represent the exchange value. The prices per unit are also used to calculate producer surplus, along with additional data on the production costs per unit, to generate the welfare value.

The System of Environmental-Economic Accounting – Ecosystem Accounting (SEEA-EA) framework provides guidance on developing 'bridging tables' that incorporate information about both exchange and welfare values (United Nations et al., 2021, p. 256)[220]. Below the guide discusses situations where it is convenient and sensible to prepare bridging tables for coastal wetlands, with some relevant examples.

A key assumption in bridging tables is that a given environmental asset provides multiple services which can be represented using exchange values and through welfare measures. For instance, if users are interested in a coastal wetland which is a crucial commercial fishing site, the value of commercial fishing is obtained from the market price of each species caught (an exchange value). The wetland could also be used for recreational fishing where value can be estimated based on the travel expenses related to the recreational trips (a proxy for exchange values). However, such recreational valuation does not account the benefits recreational fishers have gained as consumer surplus since they would be willing to pay more for the service.

Hence, it is relevant to add the consumer surplus based on estimates from a non-market valuation approach such as a travel cost model or willingness to pay from a choice experiment survey. Furthermore, other people who do not have the access nor the intention to use resources from the coastal wetland have non-use values. For example, the prospect of making the wetland available for future generations or deriving satisfaction from its existence. Therefore, the

bridging table could provide a clear understanding of all the components of the values generated from conservation of the coastal wetland to be integrated into EEA.

Other economic indicators

In addition to exchange values, SEEA-EA also compiles other measures of economic activity such as employment and increase in productivity at sub-national level, while recognizing these are not formal measures of value.

Project proponents often argue that projects will generate additional benefits through economic multiplier effects as the benefits of economic activity flow through the economy. Macro-economic general equilibrium models include such 'multiplier' effects. In ecosystem accounting, there is also interest in understanding the significance of the relationship between economic dependence on ecosystems and standard measures of economic activity, such as GDP. For example, when measures of agricultural production depend on crop pollination ecosystem services, the dependency measures could focus on the direct impact (e.g. GDP 'at risk' in the absence of the pollination service) but may also take indirect (or supply chain) effects into account by measuring multiplier effects within the economy, using the extended supply and use table (United Nations et al. 2021, p. 326)[220].

If multiplier effects are to be measured, an example of how to estimate them is given by the NSW Treasury employment calculator[242].

NSW Treasury developed an employment calculator that presents input-output (IO) employment multipliers for all industries. These employment multipliers are designed to assist agencies with:

- Automated calculation and reporting of employment estimates;
- Consistent framework application, source data and reporting;
- Understanding the link between Government expenditure, to industry impacts and eventual employment supported

Multiplier effects may be useful to report with respect to social policy objectives, but are generally not a relevant measure of economic benefit. For example, unless there is high unemployment in the economy, most projects will create different – rather than additional – jobs. In such an example, a project may shift job opportunities to a regional area to support social policies on regional employment, but only to the detriment of other areas. Welfare analyses such as benefit-cost analysis, discussed further below, often ignore multiplier effects given that they are not always a direct benefit (or cost) that results from a specific project.

Data sources

Primary data requirements for recreational and non-use values will be diverse and depend on the nature of the benefits being generated at the site.

Recreational visitation data

With respect to recreational-based services 'use' values arise from directly visiting a site. The motivation for visiting a site may differ between people, and it is acknowledged that multiple benefits may co-exist. For example, the benefits associated with recreational fishing may include elements of improved health and wellbeing, and it may be difficult to disaggregate them. However, the act of visitation provides an insight into the aggregate value being generated.

Data requirements for identifying exchange, welfare and other economic activity measures for recreational services requires primary data on the aggregate level of visitation. This can be through understanding visitation rates by either individuals or from regions, and expenditure (which can be used as an input to estimate the monetary exchange and welfare values through travel cost models). This information can be generated from

[242] NSW Treasury (2020). New South Wales (NSW) Treasury Employment Calculator –User Guide https://www.treasury.nsw.gov.au/sites/default/files/2020-10/CEE%20IO%20User%20Guide%20-%20131020%20%28Web%20Final%29.pdf

visitation surveys. If the resources are available, collection of this primary data is recommended. It is possible to collect data only on visitation rates, or jointly on visitation and expenditure. If expenditure data is not collected, the discussion about 'Valuation data (monetary values)' below should be consulted.

The cost of sourcing such data will vary widely (e.g. from a few thousand dollars to tens of thousands) depending on numerous factors including how remote and how frequently visited a site is, how heterogenous the visitor demographics are, and how complex the survey is (e.g. if including expenditure/travel cost or other non-market valuation approaches as discussed further below). Noting these costs may be prohibitive if available resources are prioritised for on-ground works rather than reporting, collection of primary visitation data might not be feasible for project-level assessment of benefits.

If it is not possible to undertake primary survey works, while it may be possible to use benefit transfer of values per trip to equivalent recreational sites to inform estimation of an economic value, an estimate of aggregate visitation will still be required to apply that value to. This could be provided, for example, through other visitation data collected through surveys in the local region, placing CCTV cameras at the site to record visitation, or recorded observations of visitation by site managers.

It is sometimes also possible to source secondary data about recreational activities from national or state-wide surveys Table 12.1. The most common challenge in using such data sources is that it may only be available at large spatial scales and be difficult to convert into smaller scales to match the specific restoration site.

The last resort when the above methods are not appliable is to use fast data curation methods such as using key informants, for example through interviews with relevant stakeholders or experts.

In cases when visitation metrics are not able to be provided or estimated, different metrics including potential visitation, predicted visitation, and other measures based on subjective indicators (e.g. density of social media posts) can be used as proxy measures for perceived recreational site quality[243,244]. 'Potential visitation' is based on an estimated maximum distance for a population within the recreational service area while 'predicted visitation' uses characteristics of the location and respondents to predict visitation rates to estimate asset values. The density of social media posts based on the locations of photographs can be used to estimate visitation rates at recreational sites[245].

[243] Barton D.N., et al. (2019). Discussion paper 10: Recreation services from ecosystems. Paper submitted to the Expert Meeting on Advancing the Measurement of Ecosystem Services for Ecosystem Accounting, New York, 22-24 January 2019 and subsequently revised. Version of 25 March 2019

[244] Wood, S.A., et al. (2013). Using social media to quantify nature-based tourism and recreation. Sci Rep 3, 2976 (2013). https://doi.org/10.1038/srep02976

[245] Scholte, Samantha S. K., et al. "Integrating socio-cultural perspectives into ecosystem service valuation: A review of concepts and methods." Ecological Economics 114 (2015): 67-78.

Table 12.1: Examples of data sources for visitation rates of recreational services.

Ecosystem services	Data source	Related studies
Recreational fishing	New South Wales State-wide or national Recreational Fishing Surveys https://www.dpi.nsw.gov.au/fishing/recreational/rfmp/research-surveys	Murphy et al. (2022)[246]
Recreational fishing	Queensland State-wide Recreational Fishing Surveys: https://www.daf.qld.gov.au/business-priorities/fisheries/monitoring-research/monitoring-reporting/statewide-recreational-fishing-surveys	Teixeira et al. (2020)[247]
Recreational fishing	Western Australia State-wide Recreational Fishing Surveys: https://www.fish.wa.gov.au/Fishing-and-Aquaculture/Recreational-Fishing/Pages/Western-Australian-Recreational-Boat-Fishing-Survey.aspx	Ryan et al. (2022)[248]
Birdwatching	Survey data and published birdwatching lists on eBird, a citizen science platform: https://ebird.org/	Carnell et al. (2019)[249]

Valuation data (Monetary values)

Primary data collected using appropriate non-market valuation methods is recommended to estimate the monetary values associated with cultural services, including recreation and existence values. This will generally mean implementing survey-based methods. The same challenges related to cost of undertaking recreational visitation surveys discussed above apply here also (with robust non-market valuation studies costing in the 10's, or even 100's of thousands of dollars depending on complexity of the site, the range of services to be valued, and accessibility of sample), so when primary data collection is not feasible the alternative is to use a "value transfer" from available literature[250,251] .

Benefit transfer for cultural services requires collection of secondary data on values of ecosystem services from various sources using systematic literature search of peer reviewed papers (primary case studies, meta-analysis), research reports and valuation tools/databases. Table 12.2 provides an example of such data sources at national or subnational level and related literature (reviewed papers and reports) for recreational services.

There are several databases that provide a searchable storehouse of empirical studies on the economic value of environmental assets and human benefits derived from ecosystem services. These may provide an alternative to conducting a

[246] Murphy, J. J., et al. (2022). Survey of recreational fishing in NSW, 2019/20 – Key Results. Fisheries Final Report Series No. 161. New South Wales Department of Primary Industries

[247] Teixeira, D., et al. (2020). 2019–20 state-wide recreational fishing survey. Fisheries Queensland, Department of Agriculture and Fisheries

[248] Ryan KL, et al. (2022). Boat-based recreational fishing in Western Australia 2020/21. Fisheries Research Report No. 327 Department of Primary Industries and Regional Development, Western Australia. 221pp.

[249] Carnell, P. E., et al. (2019). Mapping Ocean Wealth Australia: The value of coastal wetlands to people and nature. The Nature Conservancy, Melbourne.

[250] Johnston, R. J., et al. (Eds.). (2015). Benefit Transfer of Environmental and Resource Values: A Guide for Researchers and Practitioners. Dordrecht, Netherlands, Springer.

[251] Boyle, K. J., & Wooldridge, J. M. (2018). Understanding Error Structures and Exploiting Panel Data in Meta-analytic Benefit Transfers. Environmental and Resource Economics, 69(3), 609–635. https://doi.org/10.1007/s10640-017-0211-y

more comprehensive systematic literature search, but practitioners should consider how current and relevant the database is for their needs. For example, some databases may not be updated regularly to include more recent studies or may focus on studies in a different geographical region or on a specific environmental context. Database examples include:

- The Economics of Ecosystems and Biodiversity (TEEB).
- Ecosystem Services Valuation Database (ESVD)[252].
- Environmental Valuation Reference Inventory (EVRI)[253].
- Environmental Valuation Database (ENvalue).
- National Ocean Economics Program (NOEP) Non-Market database[254].
- The Value Tool for Natural Hazards[255].
- The INFFEWS Value Tool[256].

Table 12.2: Examples of data sources of monetary values for recreational related services.

Value data	Ecosystem services	Data source	Related studies
Travel cost data (e.g. fuel cost)	**Recreational fishing**	National/Statewide Recreation expenditure surveys: e.g. Expenditure on Recreational Fishing in NSW; https://www.dpi.nsw.gov.au/__data/assets/pdf_file/0009/499302/UOW-statewide-economic-survey-final-report.pdf	McIlgorm and Pepperell (2013)[257]
	Recreational fishing	Queensland State-wide Recreational Fishing Surveys; https://www.daf.qld.gov.au/business-priorities/fisheries/monitoring-research/monitoring-reporting/statewide-recreational-fishing-surveys	Kandulu et al. (2021)[258]
Consumption expenditure	**Recreational fishing**	Australian National Accounts: Tourism Satellite Account tourism https://www.abs.gov.au/statistics/economy/national-accounts/australian-national-accounts-tourism-satellite-account	
Total trip expenditure	**Birdwatching**	Tourism Research Australia (TRA) National Visitor Survey (NVS)https://www.tra.gov.au/domestic/domestic-tourism-results	Steven (2022)[259]

[252] Ecosystem Services Valuation Database: https://www.esvd.net/

[253] Environmental Valuation Reference Inventory: https://evri.ca/en

[254] National Ocean Economics Program: https://www.oceaneconomics.org/nonmarket/

[255] Value Tool for Natural Hazards (2021), Version 3.0, Bushfire and Natural Hazards CRC website, available from http://tools.bnhcrc.com.au/wtp/home

[256] CRC for Water Sensitive Cities: https://watersensitivecities.org.au/investment-framework-for-economics-of-water-sensitive-cities-inffews-value-tool/

[257] McIlgorm, A. & Pepperell, J. (2013). Developing a cost-effective statewide expenditure survey method to measure the economic contribution of the recreational fishing sector in NSW in 2012. A report to the NSW Recreational Fishing Trust, NSW Department of Primary Industries. Produced by the Australian National Centre for Ocean Resources and Security (ANCORS). University of Wollongong.

[258] Kandulu, J., et al. (2021). Economic contribution of recreational fishing by Queenslanders to Queensland: A Report for Fisheries Queensland. Fisheries Queensland

[259] Steven, R. (2022). Bird and Nature Tourism in Australia. KBAs in Danger Case Study Report. Report prepared for BirdLife Australia. Carlton, Australia. 10.13140/RG.2.2.26540.54406

Key assumptions or limitations

There are a number of key considerations relating to the valuation of cultural services.

The focus of SEEA-EA is measuring use value ecosystem services using exchange values. While non-use values are equally important to be accounted for in decision making, current guidance does not allow for these values to be reflected in exchange values. For instance, existence values for coastal wetlands are likely to be important, and sometimes large, and people will have an increase in welfare because of the restoration measures. It is therefore likely to be important to consider economic valuation and welfare values in addition to exchange values if existence values have the potential to be significant for a project site. Welfare values can be reported in bridging tables and can be utilised in economic valuation as discussed in Appendix 2.

Cultural services are measured through non-market valuation approaches originally developed to estimate welfare values. A challenge for the application of all non-market valuation methods is the distinction between exchange values and welfare values, noting that not all valuation approaches generate exchange values.

Methods exist to modify recreational demand models estimated through non-market valuation so that they can simulate an exchange value[260]. Further, it is possible for some forms of discrete choice models to be implemented so that use-related welfare values can be interpreted as exchange values, particularly where the marginal changes being valued in trade-offs in the choice experiment are at an appropriate scale to reflect a 'per unit' exchange. However, while some non-market valuation methods will support estimation of exchange values, not all of the available literature will be able to, and this further limits the ability to find suitable studies to extrapolate from when benefit transfer is required.

The resources required, in terms of expertise, time and funding, to collect primary non-market value data will inevitably mean there will be a reliance on benefit transfer to inform many project-level evaluations. The guide cautions that benefit transfer should be applied conservatively and cautiously, following best practice guidance available in the literature and with all assumptions made during the transfer process to be reported with absolute transparency.

For recreational services, there is often not only a lack of existing primary data for the non-market values of specific coastal wetland sites, but also on the visitation rates for the site that are required to aggregate values per recreational visit. The availability of visitation numbers is more dependent on primary data: there are not sufficient studies to 'transfer' visitation use associated with a site, unless models have been closely aligned (i.e. local site visitation models that can be used to infer use based on ecological characteristics of the new site). The absence of site-specific visitation data makes aggregation challenging, but also means there will be an absence of data about the socio-demographics of visitors, which further reduces the ability to make appropriate adjustments of secondary data used in benefit transfer, leading to reduced accuracy.

A further challenge is to ensure that the quantum of change is appropriate for the analysis being undertaken. This requires identifying the value of the stock of the environmental asset and applying that to changes in the stock that may occur at a site, using an exchange value. This requires an evaluation of the additionality associated with restoration of a site. Additionality in values will depend on an appropriate identification of the additionality in the biophysical system.

Uncertainties

The uncertainty that will be associated with physical and monetary account values for cultural services will be highly variable from one project to another:

[260] Campos, P., et al. (2019). Bridging the Gap Between National and Ecosystem Accounting Application in Andalusian Forests, Spain. Ecological Economics, 157, 218-236. https://doi.org/https://doi.org/10.1016/j.ecolecon.2018.11.017

- Greater confidence in primary data collected using preferred methods for valuation, relative to having to rely on values estimated using proxy methods such as benefit transfer.

- Confidence in the visitation rates or biophysical quantities used to inform any coastal valuation exercise carry through from physical to monetary accounts for a service. So, the above point applies to both physical and monetary account preparation, and uncertainty will be amplified if benefit transfer must be used for both.

- Greater confidence in primary and secondary methods for sites that are well-understood, well-studied and familiar with recreational and other community stakeholders, as awareness and understanding can reduce the variance (or randomness) in people's valuations of a site.

- Larger sample sizes (e.g. for the number of people sampled in a survey, or for the number of relevant secondary studies that can be used in a meta-regression benefit transfer) will typically make estimation of values more precise.

- The quality of a valuation exercise (which may be dictated by the sample size available but also by the budget available, time constraints, and experience of the practitioner) can affect the robustness of data collected and values estimated. This means we are likely to have greater confidence in a thorough primary data (large sample, best practice design and analysis) valuation, relative to a budget-constrained valuation.

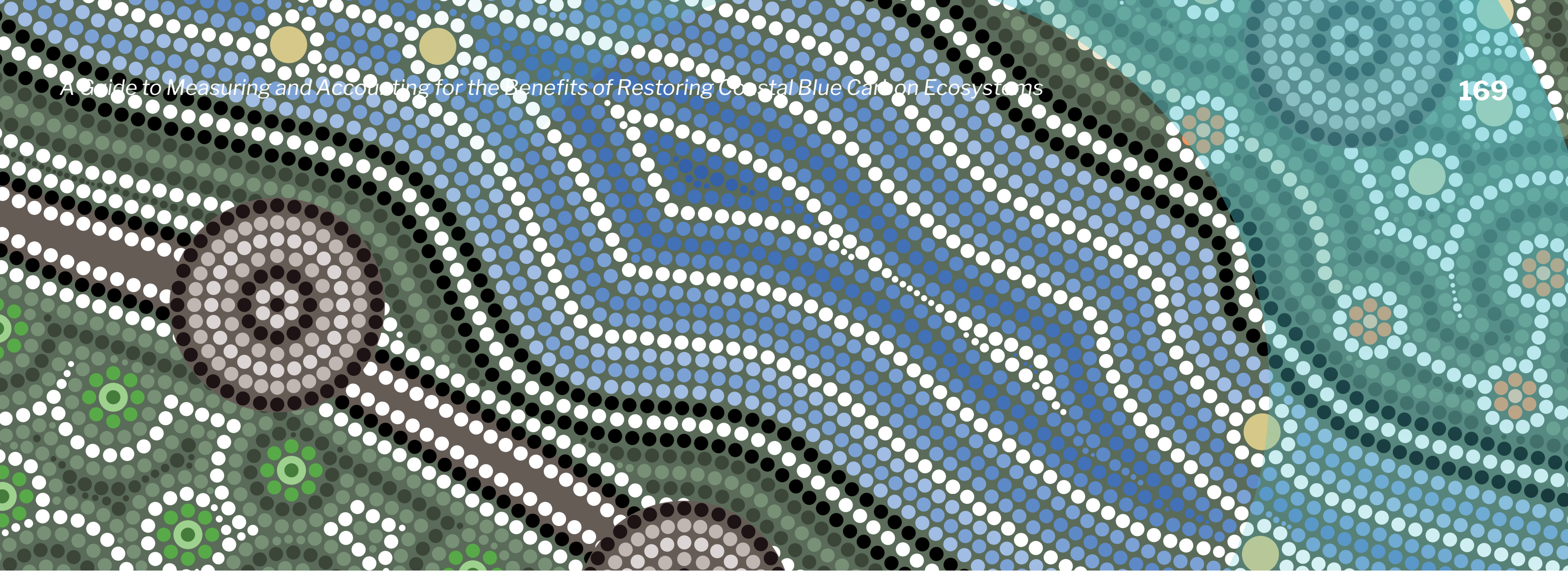
- It is not straightforward to determine whether a good-quality benefit transfer may deliver better confidence in values estimated than a poor-quality primary valuation, as this may depend on the availability of other studies that are well-matched to the restoration site.

It is important to describe how accurate the inputs are in the accounts tables, as numbers can otherwise be interpreted too liberally. Given the paucity of context and site-specific primary data available to inform SEEA-EA for coastal restoration projects, practitioners are cautioned against ignoring the need to clearly discuss (un)certainty. Reliance on benefit transfer methods implies that confidence intervals are likely to be wide for many numbers reported in cultural service monetary accounts. Care must especially be taken if trying to interpret trends over time when this may be the case, as what may look like a positive or declining trend in service provision could easily be the opposite (or neither).

Where primary valuation data is collected or meta-regression methods are used, modelling to estimate values may provide statistical measures of confidence that should be included in the SEEA-EA reporting.

However, at a practical level, it is easy to see that statistical measures of confidence will not be readily available. In these cases, we recommend the use of qualitative indicators of certainty. Practitioners can propose their own qualitative index as required, with the main criteria being that the index is reported transparently. We would recommend the index considers the issues depicted in the dot points above regarding whether the preferred method for estimating exchange (or welfare, where relevant) values was used, including whether primary or secondary data was available, sample size and other factors that define best-practice application of the method being applied. For example, Eger et al.[261] scored economic ecosystem service values associated with the Great Southern Reef based on an evaluation of 6 criteria, leading to assessments that ranged from a "High" degree of certainty in the values, to unusable (i.e. worse than very poor).

[261] Eger, A.M, Bennett, S., Zimmerhackel, J., Rogers, A., Burton, M., Filbee-Dexter, K., Wernberg, T., Gacutan, J., Milligan, B., Vergés, A. (2022) Quantifying the ecosystem services of the Great Southern Reef. Report to the National Environmental Science Program. University of New South Wales.



13. Cultural services: First Nations values

Prof. Melissa Nursey-Bray, Dr Celeste Hill, Dr Nina Wootton, Dewayne Mundraby, Dale Mundraby

13.1 Summary of section

This chapter presents a suggested approach to the development of a cultural account which can be used to complement/work alongside wider environmental economic accounting processes.

Establishing a way to account for Indigenous cultural values and uses within an environmental economic accounting (hereafter EEA) process brings unique challenges and opportunities[262]. One challenge is that Indigenous worldviews are holistic thus their relationships with the environment are not reducible to a use or service per se and their values are relational. Yet the value of building multi-faceted biocultural approaches is important not just in restoring ecosystems but in embedding and accounting for equitable societal outcomes[263]. While Indigenous knowledge systems provide opportunities to build ecosystem services and management[264], the analysis of cultural ecosystem services can also help document links to landscape, heritage and identity[265].

EEA processes rely on technocratic approaches to socio-ecological systems that presume all components are identifiable, discrete, material and hence measurable[266]. Further, models for SEEA

[262] Manero, Ana, et al. (2022). A systematic literature review of non-market valuation of Indigenous peoples' values: Current knowledge, best-practice and framing questions for future research. Ecosystem Services 54,101417.

[263] Morishige, Kanoe'ulalani, et al. (2018). Nā Kilo ʻĀina: Visions of biocultural restoration through indigenous relationships between people and place." Sustainability 10(10), 3368.

[264] Pyke, Michelle L., et al. (2018). Wetlands need people a framework for understanding and promoting Australian indigenous wetland management. Ecology and Society 23(3), 43.

[265] Tengberg, Anna, et al. (2012). Cultural ecosystem services provided by landscapes: Assessment of heritage values and identity. Ecosystem Services 2, 14-26.

[266] Reid, J. & Rout, M. (2018). Can sustainability auditing be indigenized?" Agriculture and Human Values 35(2), 283-94.; Bostedt, G.& Tommy Lundgren, T. (2010). Accounting for cultural heritage — A theoretical and empirical exploration with focus on Swedish reindeer husbandry. Ecological Economics 69(3), 651-57.

do not explicitly recognize Indigenous benefits and services[267]. It is therefore problematic and possibly culturally unacceptable to separate – and quantitatively measure - values from or traded off from each other – it is not possible to measure what is considered in Indigenous terms, the unmeasurable. There is also no substitute for sacred goods and services. Indigenous knowledge is specific and culturally held by certain people, so how it gets treated within an EEA process needs care.

Assessments need also to respect and reflect recognition of various cultural losses that may have occurred in the area due to colonization[268]. The inherent variability in Country-based value systems means a common EEA assessment process may not be appropriate[269], and different populations may hold different preferences/values around/for the benefits of the system. For example, Indigenous peoples that still live on Country may have different views to those that live outside it but who still affiliate with it and their group. The heterogeneities amongst Indigenous groups may pose challenges in aggregating responses, and the communal property rights amongst some Indigenous groups preclude individual utility structures.

Another consideration of cultural accounting at restoration sites is the need to understand that it may not be possible to split up values between those within the restoration site and more broadly across Country. Given the holistic nature of Country, asking First Nations owners to separate off and describe services for a particular area may not be possible as there may be services that were pertinent to the country as a whole. This can create some blurring in how services are articulated. Thus, in attempting to separate services within a bounded site, which is included within but not *all* of Indigenous Country, it is harder to gauge the extent and value of those services.

However, the identification of Indigenous values in EEA process can have benefits. It can assist in Caring for Country for the relevant Indigenous group but also identify the impact /value of Indigenous cultural resource management (ICNRM) on / for the system . Further, current EEA processes tend to focus on the flow of benefits from nature to people but do not recognize the reciprocal responsibilities of people to care for the environment, enacted by Australian Indigenous peoples via the process of Caring for Country. Cultural accounts can also help to document biocultural values in formats relevant to management (ibid).

Acknowledgement of this circularity is integral to developing a cultural account within the SEEA process.

For example, the interconnectedness of Indigenous connection to and caring for Country suggests a need to develop circular rather than linear modes of gathering information in developing a cultural account[271]. Aligned with this approach it is important to build in the complementary concept of people's contribution to nature as being as important to a cultural account as what nature/ecosystems can offer to people[272]. The relationship between nature's contributions to people and people's contribution to nature is thus recognized explicitly and emphasises the circular and holistic nature of interconnection, which is contra distinct to the linear and atomistic character of most accounting models.

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[267] Normyle, Anna, et al. (2022). Ecosystem accounting and the need to recognise Indigenous perspectives. Humanities and Social Sciences Communications 9(1), 133.

[268] Duffield, J. W., et al. (2019). Natural resource valuation with a tribal perspective: a case study of the Penobscot Nation. Applied Economics 51(22), 2377-89.

[269] Sangha, K. K., et al. (2017). Challenges for valuing ecosystem services from an Indigenous estate in northern Australia. Ecosystem Services 25, 167-78.

[270] Larson, S., et al. (2023). Piecemeal stewardship activities miss numerous social and environmental benefits associated with culturally appropriate ways of caring for Country. Journal of Environmental Management 326, 116750.

[271] Larson, S. et al. (2023). Piecemeal stewardship activities miss numerous social and environmental benefits associated with culturally appropriate ways of caring for Country. Journal of Environmental Management 326, 116750.

[272] Matuk, F. A., et al. (2020). Allying knowledge integration and co-production for knowledge legitimacy and usability: The Amazonian SISA policy and the Kaxinawá Indigenous people case. Environmental Science & Policy 112, 1-9.

Ultimately, to obtain reliable research outputs relating to the cultural values and services of the restoration site, worldviews need to be acknowledged. External researchers work from the worldview implicit in the environmental economic accounting approach, which holds the site as a distinct or bounded part of the landscape, to which service provisions can be attributed. This contrasts with the indigenous worldview, which does not delineate between the site and wider landscape, particularly when it comes to the value and meaning of that landscape.

The inclusion of sociocultural valuation techniques, combined with knowledge gained via other EEA processes, enables the development of policies and programs that can build/protect blue carbon ecosystems while acknowledging they are also cultural domains[273].

Taking into consideration all these factors, the following approach is suggested to undertake the development of a cultural account. To undertake the most effective cultural accounting process, two procedures need to occur: (i) an engagement and partnership process and (ii) development of the cultural account itself. The following five step approach illustrated below (Figure 13.1) is suggested that will provide a pathway by which to implement both procedures.

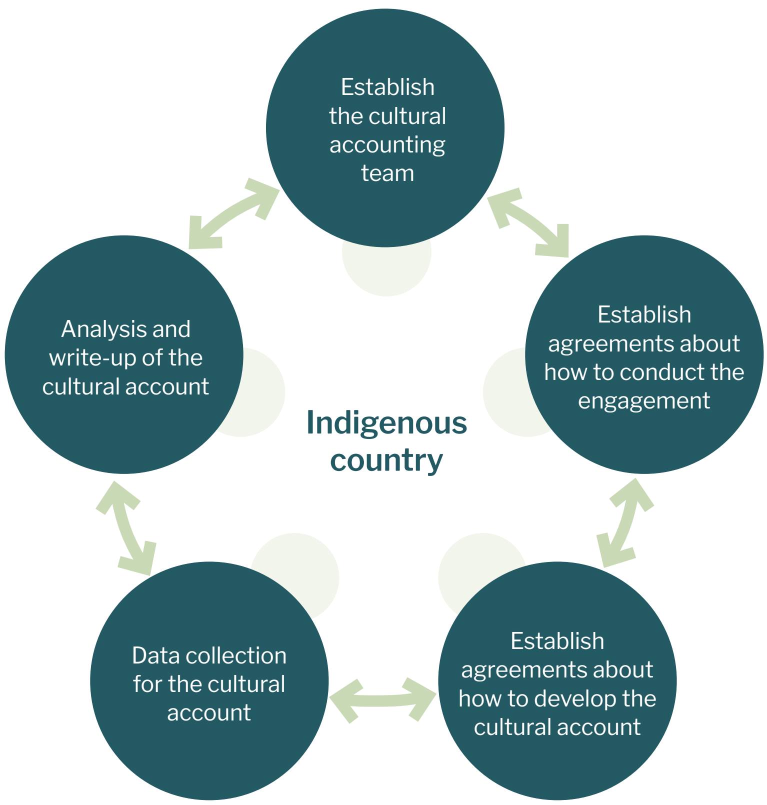


Figure 13.1: Five-step pathway for cultural account process.

[273] Scholte, S. S. K., et al. (2015). Integrating socio-cultural perspectives into ecosystem service valuation: A review of concepts and methods. Ecological Economics 114, 67-78.

13.2 Detailed section

Methods

Step 1: Establish the cultural accounting team

Working with Indigenous communities is central to any account as every place in Australia that could be identified for accounting, will also be on Indigenous Country. Therefore, the first step in any cultural account is to determine who the team will be. This is identified as the first step because best practice cultural accounting is a process that will be co-designed and co-led with Indigenous partners. It is also the first step because it will enable the establishment of appropriate engagement at the very outset.

To assemble the best and most culturally appropriate team, groundwork is needed to identify the most appropriate people or groups to approach. If the 'right' people are not appointed at the very beginning, the entire account is compromised. The creation of a strong team will require identification of the following: (i) the history of Country and impact of colonization in the region, (ii) identification of who the native title holders are and (iii) building a data base of all the Indigenous groups, bodies, agencies, and key individuals in the area.

While appointment of Indigenous leads to develop the account and undertake engagement with relevant Indigenous representatives is ideal, it will not always be possible. In this case, it is strongly recommended that if a non-Indigenous cultural lead is appointed that they have extensive prior experience working in partnership with Indigenous people. This will ensure the process of engagement and accounting can be expedited in an efficient but culturally appropriate manner.

Step 2: Establish agreements about how to conduct the engagement

Engaging with the Indigenous representatives and owners of the Country on which the account is occurring is the most important step in the account process. Without effective partnerships development of the account will not be possible.

Governance: Establishing a cultural advisory function from the very beginning is vital as is the adoption of Indigenous engagement principles. Appointment of someone with expertise in Indigenous engagement and environmental management is another core principle. Provision of enough time and resources to do this accounting properly is essential.

Written agreements: Once engaged with the First Nations' group, and they have agreed to be cultural partners, the next stage is to collaboratively negotiate the terms of the engagement. This may be via a range of mechanisms, but it is recommended that there be some form of written agreement. This can be in the form of a Memorandum of Understanding (MOU) or via a written protocol but needs to occur in addition to the any institutional ethics approvals that may need to be obtained (see Table 19). The agreement should outline agreements and understandings about the following:

- **i.** Who needs to be engaged;
- **ii.** Identify the purpose of the EEA and questions to be asked;
- **iii.** How will the co-designing and cultural input happen;
- **iv.** What are the benefits of the process;
- **v.** Identify the appropriate intellectual property frameworks;
- **vi.** Identify which and whose values are to be considered;
- **vii.** Identify and agree on the sources of information to be used and how/when they will be collected;
- **viii.** Agree on how and when cultural reviews will occur;
- **ix.** Agree on how the information will be disseminated/published; and
- **x.** Identify any key limitations.

Any written agreement needs to be underpinned by an acknowledgement of Indigenous conceptualizations of the system within the EEA.

Communications and engagement: Throughout the project communications should be ongoing. Making sure that the Indigenous partners are involved and aware of every stage of the project, including the wider one is built into this process. Ensure that cultural review processes are undertaken. At the end of the project, Indigenous parties should be included in the communications about the results of the wider EEA and the articulation of its co-benefits.

Resources: There are many resources available that provide a guide for how to engage with Indigenous partners appropriately. One of these is the Code of Ethics by the Australian Institute of Aboriginal and Torres Strait Islander Studies (AIATSIS). This guide provides the information required to undertake effective engagement in research (Table 13.1).

Table 13.1: **Outline of the Australian Institute of Aboriginal and Torres Strait Islander Studies Code of Ethics. For more information see: Code of Ethics | AIATSIS.**

Australian Institute of Aboriginal and Torres Strait Islander Studies Code of Ethics
The AITSIS Code is structured according to four principles that underpin ethical and responsible Aboriginal and Torres Strait Islander research, which are: **1.** Indigenous self-determination **2.** Indigenous leadership **3.** Impact and value **4.** Sustainability and accountability Each principle includes a set of responsibilities for conducting Aboriginal and Torres Strait Islander research.
Each principle gives rise to responsibilities that are elaborated under the following headings: • recognition and respect • engagement and collaboration • informed consent • cultural capability and learning • Indigenous led research • Indigenous perspectives and participation • Indigenous knowledge and data • benefit and reciprocity • impact and risk • Indigenous land and waters • ongoing Indigenous governance • reporting and compliance

Step 3: Establish agreements about how to develop the cultural account

Once the terms of the collaboration and engagement have been agreed, the next step is to decide how to undertake the cultural account. It is suggested as a first principle, the approach be informed by the seven key questions Manero et al. (2022)[274] suggest for non-market valuation of Indigenous people's values and which have been amended for a cultural accounting approach. These questions include:

- **i.** What is the purpose of the account?
- **ii.** How can Indigenous knowledge inform the account?
- **iii.** Who benefits from the account?
- **iv.** What ethical frameworks apply?
- **v.** Whose values are being considered?
- **vi.** What is the expected change?
- **vii.** How are the limitations of cultural accounting within the SEEA process acknowledged?

The deployment of and discussion about these questions will help inform choices about how to do the account and will be especially helpful in establishing what values are held about the system. Further, there will need to be agreements around how cultural values and services are to be documented and understood, and then how they will be represented in a series of tables.

Another decision that needs to be made at this point, is how to incorporate the economic services that Indigenous people may derive from the ecosystem. For example, they may run a tour or get financial advantage from the area in some way. Within a SEEA Framework, it is possible to document the economic benefits of the ecosystem services in the cultural account, or this information may also be represented in the socio-economic account. The important thing is to ensure that it is not counted twice.

Step 4: Data Collection for the cultural account

Data collection for the development of a cultural account needs to be undertaken so that enough information is collected to map the relationship between cultural values of the site and the ecosystem benefits the Indigenous peoples derive from that site.

It is the mapping of the relationship between the two that enables identification of the ecosystems services the area provides. Understanding the values attributed to the region also facilitates an understanding of what value the ecosystem receives from the people, via caring for Country.

Documenting Values: In the determination of how values are to be defined and which ones to assess, a tailored approach is required. At the outset, there is a need to define what is understood by a 'cultural value'. Cultural values may be direct and indirect use values (e.g. Traditional Indigenous food, Indigenous led natural resource management for carbon sequestration), or altruistic/bequest or existence values. Spiritual values as well as health values need inclusion. A summary of how values can be conceptualized is seen in Table 13.2.

Cultural values are defined here as the importance people or groups assign to bundles of ecosystems and cultural services in a place/Indigenous Country (adapted from Scholte et al 2015)[275]. This includes the idea of shared values about and affiliation to Country, and whether people live within it, as this enables a meta narrative about site value that goes beyond the aggregated utilities of individuals[276].

The Millennium Ecosystem Assessment (MEA, 2005) defines cultural values as the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development,

[275] Scholte, S. S. K., et al. (2015). Integrating socio-cultural perspectives into ecosystem service valuation: A review of concepts and methods. Ecological Economics 114, 67-78.

[276] Irvine, K. N., et al. (2016). Ecosystem Services and the Idea of Shared Values. Ecosystem Services 21, 184-93.

reflection, recreation, and aesthetic experiences. Fish et al. (2016)[277] define cultural ecosystem services as the interaction between environmental spaces (i.e., physical settings such as coasts, woodlands, allotments) and the cultural or recreational practices (e.g. fishing, walking, gardening) that takes place within them. The flow of goods and services from Indigenous Country, while it may not encompass the relational and intangible values, can be translated to economic values where appropriate. Wellbeing is also suggested as a value/indicator, as it could be used as a proxy indicator to go some way to recognizing rights to access, use, enjoy, and feel part of the connected 'whole' of Country.

Co-creation of value representations is paramount. Economic value that translates to wellbeing, security and self-determination of the Traditional Owners is important, but is not the only layer of value. The value of Country - and the many dimensions of its meaning - must be acknowledged and described, using qualitative methods that account for significant non-monetary value.

At the heart of a cultural account, is the question of how best to render an authentic cultural account of the interlinked values that are afforded by the area being investigated. This question must be answered in a co-creative process with the people themselves. In this context, Section 2.4 of the SEAA – EE provides useful support for the need to incorporate multiple value perspectives, recognising that monetary valuations while important, can be limiting in development of cultural accounts. One thing is certain: value representation encompassing and going beyond the monetary, is essential. In view of these considerations, this section extends the identification of values as described above and visually represent our suggestion for encompassing non-monetary and economic Indigenous values with respect to Country (Figure 13.2). This approach emphasises the circularity and interconnectedness of cultural values ascribed to ecological services arising from an ecosystem.

There are a range of established techniques that have been used to identify socio-cultural values

Table 13.2: A summary of how values can be conceptualized, and which demonstrate material and non-material values.

Value domain	Examples of values
Physical sites that support cultural activity	Cultural /dreaming sites, food sites, medicine sites, art/knowledge sites
Natural resources that support cultural activity	Water, air, soil, plants, animals
Accessibility to the system	Level of disruption, level of revitalization, extent, scope of access to Country, native title recognition of Country
Knowledge about the system	Language, stories, TEK, Lore, Calendar
Wellbeing indicators	Residence/housing on Country, social/familiar relations, health indicators, security/work
Uses of the system	Hunting, harvesting, medicine, recreation, art and craft
Connective Benefits of the system	Spiritual, economic, environmental

[277] Fish, R., et al. (2016). Conceptualising Cultural Ecosystem Services: A Novel Framework for Research and Critical Engagement. Ecosystem Services 21, 208-17.

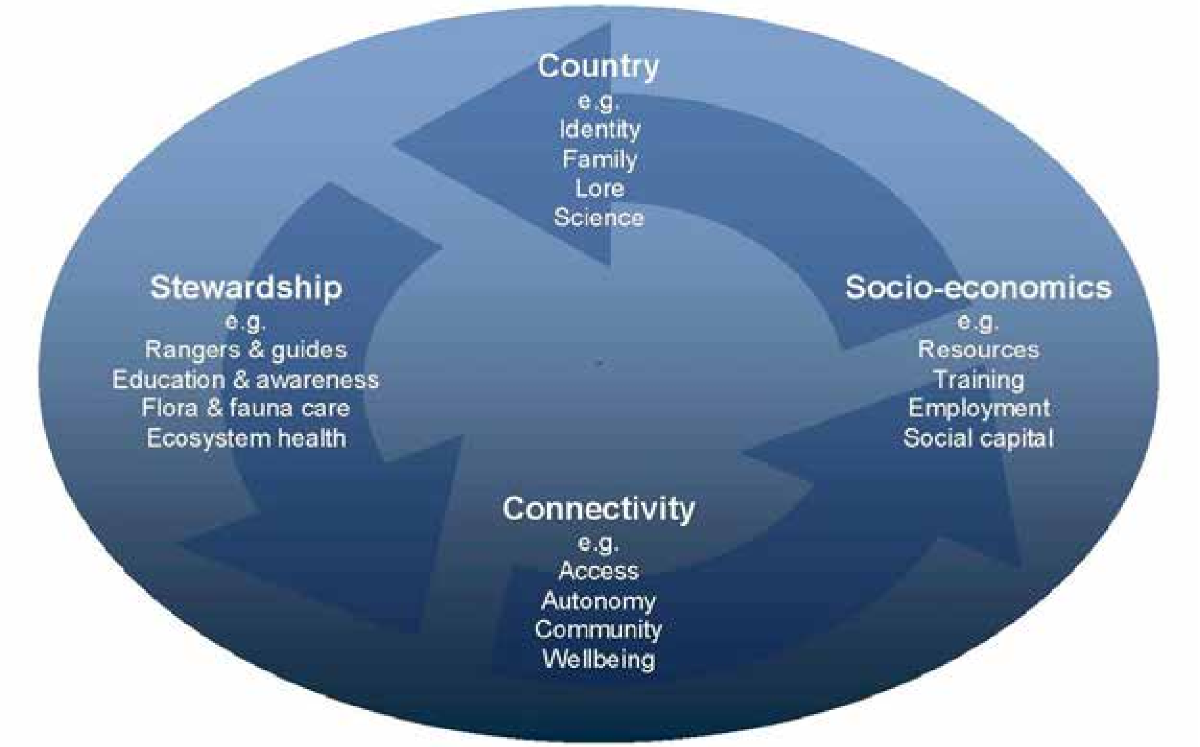


Figure 13.2: Wholistic diagram showing aspects of cultural values encompassed in 'Country'. The collaborative processes described in this guide are used to elicit, collate and illustrate the number and type of links between various aspects of value - that relate to an environment and how it changes (i.e., restoration value). Connections may be evident as actions like fauna care, or as expressions of meanings, such as descriptions of identity. Aspects of value may change across different cultures, and this schema can be altered to reflect diverse values and diverse environments.

and then their relationship to ecosystems services into EEA and ecosystem valuations. These include observation approaches, documentary research, expert-based approaches, in depth interviews, focus groups, questionnaires, photo and other mapping techniques[278]. Others have applied wellbeing indicators[279].

An Indigenous specific EEA valuation framework also needs to consider:

i. How do various ecosystems contribute to Indigenous peoples' capabilities and wellbeing?

ii. What is an appropriate framework to recognise Country-related Indigenous values for making policy decisions?

iii. What are the appropriate methods to measure the monetary values of those ecosystems' services?

Another method is to apply Sen's Capability approach which enables identification of the socio-cultural benefits for Ecological Services to enhance wellbeing[280]. An approach trialed by the UK National Ecosystem Assessment, identifies Cultural Ecosystem Services as the 'the

[278] Scholte, S. S. K., et al. (2015). Integrating socio-cultural perspectives into ecosystem service valuation: A review of concepts and methods." Ecological Economics 114, 67-78.

[279] Sangha, K. K., et al. (2017). Challenges for valuing ecosystem services from an Indigenous estate in northern Australia. Ecosystem Services 25, 167-78.

[280] Sangha, K. K., et al. (2017). Challenges for valuing ecosystem services from an Indigenous estate in northern Australia." Ecosystem Services 25, 167-78.

interactions between environmental spaces (i.e., physical localities or landscapes), and the activities that occur there'[281].

Data Collection: Based on these ideas, multiple data types can be collected with a range of methods (Table 13.3). It is important to understand that the choice of which method or methods to use will be determined by the nature of the case study and the guidance from the Indigenous partners as to what is the most culturally appropriate method to use. Generally, the use of at least 2 – 3 methods is encouraged as it enables triangulation of data, and corroboration of key themes/findings.

It is at this point that the team will contact the community and decide (i) who to consult and (ii) how to collect information about values and ecosystem benefits. At this point a series of questions will be developed that will enable identification of both cultural values and ecosystem benefits.

Examples of the types of questions that might be asked are given below (Table 13.4).

Table 13.3: Summary of types of methods that can be used for data collection in the development of a cultural account.

Desktop review: data points	Participatory engagements and knowledge co-production
Archival history material: archives can be located within libraries, newspaper archives, church archives, birth and death records, copies of personal letters, media entries. These materials can be collected, read, analysed using qualitative data analysis and historical techniques to reveal findings and insights for incorporation into the account.	**On Country workshops**: Workshops that involve many Indigenous groups in the area can be held on Country and offer people the opportunity to be in their place and share Country with others.
Church/mission records: Given the history of colonization and therefore missionization of Indigenous peoples in Australia, it is always useful to approach local, state and national church groups such as the Anglican, Lutheran, Catholic and others and request permission to access their records. These materials can be collected, read, analysed using qualitative and historical data analysis techniques to reveal findings and insights for incorporation into the account.	**Focus groups**: Focus groups offer the opportunity for getting deeper information on specific issues. They can also be used to work with women and men separately where cultural mores need this to happen, or to engage with particular clan groups within a larger people/Country or with particular people who may have responsibility for specific parts of Country/are the stewards for it.
Literature: published and grey literature about the Indigenous group that you wish to work with will be another appropriate source of information. These materials can be collected, read, analysed using qualitative data analysis and review techniques to reveal findings and insights for incorporation into the account.	**Cultural Value and Site Mapping**: This is a process whereby Indigenous owners and research team work together to identify key sites of significance, their values and uses and ultimately produce a map that reflects this information.
Museum Records: Museums will often have, both within their exhibitions, but also the history of their minutes, past exhibitions, artefacts, correspondence, photos, etc - information about the cultural use, cultural practices and sites of local Indigenous groups. These materials can be collected, read, analysed using qualitative data and historical analysis techniques to reveal findings and insights for incorporation into the account.	**Semi structured interviews**: Can be deployed to gather in depth information about sites, used, values, to build individualized cultural accounts which can then be used to build a bigger picture of the overall cultural account. Interviews also offer the opportunity to make linkages between cultural and many other uses/accounts.

[281] Bryce, R., et al. (2016). Subjective Well-Being Indicators for Large-Scale Assessment of Cultural Ecosystem Services. Ecosystem Services 21, 258-69.

Table 13.3: cont.

Desktop review: data points	Participatory engagements and knowledge co-production
Local and other government records: local historical government records will include information about past employment, births and deaths, receipts for payments, services offered, education records, tenures (and changes in), health, infrastructure, transport, and other relevant evidence that can be used to build a cultural account for the area. These materials can be collected, read, analysed using qualitative data analysis techniques to reveal findings and insights for incorporation into the account.	**Story telling**: One way of gathering information about sites and their value is to undertake a storytelling or 'yarning' methodology. It is a method that allows the free form narration of stories, and histories and connections to Country in a way that emerges as a series of stories about connection, elicited by 'yarning' (open but directed conversation about a given topic), and which assist in building a cultural account.
Media and local group records: media accounts from newspapers, radio transcripts, local group newsletters, minutes, and many other sources will yield information about cultural sites, events, practices that can assist build a cultural account. These materials can be collected, read, analysed using qualitative data nd media analysis techniques to reveal findings and insights for incorporation into the account.	**Field trips**: Going out together on Country is a very effective way of both seeing the sites but also gathering information about the area first-hand. It is also an effective method for documenting change to the sites over time.
Anthropological and archaeological records including maps: over the last 200 or so years many anthropologist and archaeologists have interacted with and written detailed accounts of the local Indigenous groups. Information may include mapping of sites, description of the people, their culture and practice, engagement with sites, documentation of language. All of these are important sources for review of what and how the sites were used and valued. These materials can be collected, read, analysed using qualitative data nd media analysis techniques to reveal findings and insights for incorporation into the account.	**Photo voice**: Photo voice is a very visual and oral means by which Indigenous partners can be trained to undertake some recording of cultural accounts in their own time, and by which often, access to a wider range of people is enabled – as those Indigenous researchers connect to their own family and friends. It is an effective way of getting localized information about sites and the people.
Published cultural information: Many Indigenous ranger, Elder, Knowledge and other cultural groups may have been publishing cultural information in order to preserve and protect it. This knowledge may be kept and held internally or published on a web site – seeking access to or consulting with Indigenous partners to understand what information is available here may yield important information/data about the site. These materials can be collected, read, analysed using qualitative data nd media analysis techniques to reveal findings and insights for incorporation into the account.	**Note**: These methods and sources of information are suggestions. The combination of methods and sources that occur will depend on the time and resources available, the expertise of the team in being able to undertake these engagements, and the specific people and location.
ABS data: The ABS holds a number of data sets that comprise key information about Indigenous peoples. This information can be found within the specific Indigenous section, as well as within breakdowns of local government demographics, as well as in the accounts for education, employment and health. A thorough analysis of what ABS data sets can reveal will offer another key source of information.	There are many articles, tools and guides that can be access that show examples of how these tools have been used in practice.

Table 13.4: Examples of questions that could be used in collecting information for a cultural account:

- Describe your role in relation to the site, and how long you have been in that role.
- Do you have any recollections of the area from before that time?
- What does this area mean to you?
- What changes in the landscape have you noticed over time?
- Have you seen any changes in plants or animals in that time?
- What do those changes mean to you?
- Have you seen any benefits from the changes?
- Have you seen any negative effects from the changes?
- Do you have any other comments or observations about the area and what it means for you?

The chosen methods will then be deployed to answer these questions. This step is the data collection phase and is the point at which the modes of engagement with the Indigenous group need to be revisited. It is at this point that face-to-face engagement with traditional owners occurs, and where the MOU or agreements about how to engage and gather information become operational.

Once the data has been collected, further decisions need to be made about how to analyse the information collected to distil key findings and results. Analysis needs to include two dimensions (i) thematic or other data analysis and (ii) cultural review.

Thematic and other analysis of the data collected

This stage will comprise discussion about what methods will be used to analyse the data gathered. This may be via qualitative methods such as thematic analysis or narrative analysis. The analysis software NVivo can also be used to develop quantitative representations of qualitative data which will achieve complementarity across the wider EEA system. Economic benefit can be calculated by using the substitute value of aspect welfare savings plus associated employment opportunity costs for people such as Indigenous Rangers and travel expenses to visit cultural sites[282].

Cultural review

Although the process of cultural accounting should be done in partnership with Indigenous owners, it is at this stage that their cultural review of the findings is of especial importance. It is essential that how Indigenous values and then ecosystem benefits are constructed align with Indigenous ways of seeing the system. The cultural review also facilitates an additional layer of analysis and verification of the results.

Step 5: Analysis and write up of account

The preparation of the account tables is the culmination of both the Indigenous engagement and the identification of values and ecosystem benefits derived from the date collection.

Development of accounts for ecosystem cultural services is difficult because the SEEA Ecosystem Accounting provides limited guidance on cultural ecosystems services. Further, ecosystem services

[282] Sangha, K. K., et al. (2017). Challenges for valuing ecosystem services from an Indigenous estate in northern Australia. Ecosystem Services 25, 167-78.

in SEEA are defined as what ecosystems provide to people and not what humans offer nature[283]. The protection and management of environment by people is covered in the SEEA Central Framework (pp 101 to 129, Section 4.3) which offers the basis of a potential accounting solution. The following account tables represent one pathway that attempts to conceptualise accounting for cultural services.

The suggested tables draw upon the definition provided by the UNESCO Framework for Cultural Statistics (2009)[284], which defines cultural services as those that satisfy cultural interests or need, and where they do not represent cultural material goods in themselves but facilitate their production and distribution. A definition provided by the FAO defines cultural services as the non-material benefits people obtain from ecosystems, which can include recreation, mental health, tourism, spiritual, aesthetic, visual and a sense of place[285] offers further detail to how to conceive cultural services. The SEEA definition of cultural services is services "which are generated from the physical settings, locations or situations that give rise to intellectual and symbolic benefits experienced by people from ecosystems through recreation, knowledge development, relaxation and spiritual reflection" and provides guidance for how to understand cultural services.

All definitions share a concern with how to articulate non-material benefits from ecosystems and acknowledge that they serve a particular purpose that may be hard to tangibly measure. The Cultural Account tables below have been prepared in alignment with these definitions. The tables thus offer a suggestion for how cultural values and services could be measured, and approximate monetary valuations. These tables can be used not just for blue systems but also for terrestrial systems.

The tables implicitly reflect cultural services for Indigenous Country, inevitably reflecting the interconnection between land and sea. This circularity, inherent in Indigenous perspectives and articulated ideas of Country (Figure 13.2) is not apparent in ecosystem accounting. The following ecosystem services account tables seek to reconcile these two elements by representing this two-way relationship as circular, and the tables reflect this circularity via use of the blue and green.

[283] Normyle, A., et al. (2022). An Indigenous Perspective on Ecosystem Accounting: Challenges and Opportunities Revealed by an Australian Case Study. Ambio 51(11), 2227-39.

[284] UNESCO. (2009)/ The 2009 UNESCO framework for cultural statistics (FCS). Montreal, Canada: United Nations Educational, Scientific, Cultural Organization.

[285] Dickinson, D. C. & Hobbs, R. J. (2017).Cultural ecosystem services: Characteristics, challenges and lessons for urban green space research." Ecosystem Services 25, 179-94; Hernández-Morcillo, M. et al., (2013). An empirical review of cultural ecosystem service indicators. Ecological Indicators 29, 434-44.

Figure 13.3: Services are circular as they are either provided for nature to benefit society or by people.

Cultural account tables

The tables below show suggested cultural condition account tables based upon the SEEA tables[286]: (i) Table 13.5 provides the definitions of cultural ecosystem services as per the SEEA augmented by additional services; (ii) Table 13.6 defines the attributes and indicators of each attribute that contribute to the Trinity Inlet cultural ecosystems services; (iii) Table 13.7 shows the interrelationships of cultural attributes and cultural services and the relative importance of each attribute to each service , and; the example account in Table 13.8 and Table 13.9 presents a cultural ecosystem services account with samples of potential outputs.

[286] Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW.

Table 13.5: Reference list of cultural ecosystem services, from Table 6.3 in SEEA guidelines (shaded), plus additional definitions of identified service[287] (non-shaded).

Cultural services	Definitions
Recreation-related services	Recreation-related services are the ecosystem contributions, in particular through the biophysical characteristics and qualities of ecosystems, that enable people to use and enjoy the environment through direct, in-situ, physical and experiential interactions with the environment. This includes services to both locals and non-locals (i.e., visitors, including tourists). Recreation-related services may also be supplied to those undertaking recreational fishing and hunting. This is a final ecosystem service.
Visual amenity services	Visual amenity services are the ecosystem contributions to local living conditions, in particular through the biophysical characteristics and qualities of ecosystems that provide sensory benefits, especially visual. This service combines with other ecosystem services, including recreation related services and noise attenuation services to underpin amenity values. This is a final ecosystem service.
Educational, scientific and research services	Education, scientific and research services are the ecosystem contributions, in particular through the biophysical characteristics and qualities of ecosystems, that enable people to use the environment through intellectual interactions with the environment. This is a final ecosystem service.
Spiritual, artistic and symbolic services	Spiritual artistic and symbolic services are the ecosystem contributions, including physical sites and biophysical characteristics and qualities of ecosystems - recognized by people for their cultural, historical, sacred of religious significance. These services may underpin people's cultural identity and may inspire people to express themselves through various artistic media. This is a final ecosystem service.
Other cultural services	
Wellbeing services	Wellbeing services are the ecosystem contributions to the specific enhanced wellbeing of local people. This includes improved wellbeing due to employment, housing, education and social benefits provided by the ecosystem. This is a final ecosystem service.
Stewardship services	Stewardship services are the set of practices that involve sustainably managing the natural resources and harvests of lands, territories, waters, and coastal seas. Stewardship services relate specifically to upholding these caring responsibilities for future generations. This is a final ecosystem service.
Provisioning Service – Wild animals, plants and other biomass provisioning services	Wild animals, plants and other biomass provisioning services are the ecosystem contributions to the growth of wild animals, plants and other biomass that are captured and harvested in uncultivated production contexts by economic units for various uses. The scope includes non-wood forest products (NWFP) and services related to hunting, trapping and bio-prospecting activities; but excludes wild fish and other natural aquatic biomass (included in previous class). This is a final ecosystem service

[287] Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW.

Table 13.6: Reference list of the attributes contributing to ecosystem services for First Nations[288]. Green text represents services provided by society to benefit nature, while blue text represents services provided by nature to benefit society.

Attribute	Definition	Indicator	Metrics/units of measurement
Knowledge	Traditional knowledge or 'lore', as handed on from preceding generations & from current – to new generations. Knowledge encompasses social rules, spiritual practices & identity, song-lines, social structure & connections (kinship). Includes knowledge of environmental cues & 'caring for Country' (see below). Knowledge also refers to any other knowledge types that TO's may use e.g.: **science** and '**western**' and '**eastern**' **philosophy**. **Knowledge** (all types) includes storage & preservation for long term maintenance of knowledge (e.g. library, museum).	**Number of TYPES of knowledge transmission, dissemination & maintenance facilitated or enhanced by the ecosystem. Measure amounts within each type where possible.** Example types: Stories, either verbal or written, regardless of medium. Songs, dances & artworks that contain knowledge about any elements as per definition. Refers to both 'in-person' and indirect transmission (e.g. online). Research outputs e.g. biological surveys, social sciences inquiry, neuroscience outputs, philosophical writings – determined by what TO's engage with / use.	**Counts of distinct thematic meanings identified in qualitative data;**
Recreation	Traditional Owners' use of the site for enjoyment / leisure / sport / entertainment activities. Activities may vary greatly according to landscape type.	**Number of TYPES of recreation activities at the site or increased by the ecosystem. Measure amounts within each type where possible.** Example types: Family gatherings, camping, recreational fishing, handcrafts.	**Counts of distinct thematic meanings identified in qualitative data;**
Harvesting	Gathering and / or making use of any organisms or abiotic resources, for food, medicinal or other cultural practices, including ceremony. **Types of organisms should be reviewed by Traditional Owners**, who may recognise (hence value) categories of organisms differently from scientific 'species'.	**Number of TYPES of organism & resources used at or increased by the ecosystem. Measure amounts within each type where possible.** Example types: Clay for body-paint, reeds for weaving, bark for shelter, herbs for medicine, meat, vegetables.	**Counts of distinct thematic meanings identified in qualitative data;**

[288] Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW.

Table 13.6: cont.

Attribute	Definition	Indicator	Metrics/units of measurement
Identity	Ways of determining 'self' at all scales, Individual & group scales. Identity includes past, present & future; includes ways of understanding past identity, understanding & reviewing current identity & developing forward-looking identity. This attribute has deep overlap with knowledge. In accounting, determine whether data focusses on a) knowledge or b) participant expression of identity.	**Number of WAYS identity** (individual & group) **is facilitated or increased by the ecosystem. Not quantifiable beyond 'ways'**. Example types: (receiving) Self: **I am** recognised by Country Group: '**we are** children of ancestral spirits' (giving) Self: **I'm a** ranger, teacher, guide. Group: '**we are** protectors, healers, stewards...'	**Counts of distinct thematic meanings identified in qualitative data;**
Access	Capacity of Traditional Owners to be on Country & access parts of Country & move around freely. Critical context: extent of historic disruption from colonisation & external factors that affect current access e.g. tenure, Native Title.	**Number of WAYS access is facilitated or enhanced by the site ecosystem. Measure amounts within each 'way' where possible.** Example ways: Increase in visits (across all purposes). Increase in land area that can be accessed. Increase in times of day / year that Country can be accessed.	**Counts of distinct thematic meanings identified in qualitative data;**
Social capital	Non-market value of education, skills training, networking, self-respect, confidence, experience, reputation & increased public profile i.e., social capital. This attribute is distinct from dollar-values like 'income' or 'market value' – instead represents social aspects of capacity development.	**Number of WAYS socio-economic capacity is facilitated or enhanced by the ecosystem. Not quantifiable beyond 'ways'**. Example ways: Gaining broader work experience, engagements with external entities, building network & profile, i.e., reputation & social capital	**Counts of distinct thematic meanings identified in qualitative data;**

Table 13.6: cont.

Attribute	Definition	Indicator	Metrics/units of measurement
Community cohesion	Factors that strengthen solidarity, cohesion & contentment shared among members of the local community. This category accommodates any influences on collective wellbeing, not already represented in others.	**Number of WAYS community cohesion is facilitated or enhanced by the ecosystem. Not quantifiable beyond 'ways'.** Example ways: Conflicts / rifts that have been remedied. Effect of events & meanings that create sense of sharing & bonding. Negative influences that have been counter-acted. Events / initiatives that improve community cohesion.	**Counts of distinct thematic meanings identified in qualitative data;**
Caring for Country	Country is a living entity, of which the people are part. 'Caring for Country' includes any actions to maintain the health and wellness of Country, whether those actions relate to physical features or spiritual features of Country. (Knowledge of such is accounted for under 'knowledge' attribute.)	**Number of ACTIONS considered 'caring for Country' by TOs - facilitated or increased by the site. Not quantifiable beyond 'actions'.** For example: Spiritual: the right people visiting the right places at the right times. Appropriate communications with ancestors or totemic species. Physical: protecting Country from inappropriate harvesting & vandalism. Risk management via controlled burns & control of invasive species.	**Counts of distinct thematic meanings identified in qualitative data;**

Table 13.7: Matrix table showing interconnection of services where one attribute is relevant across a number of services. y[289]. A different way of conceptualising services, where instead of being atomistic, and having all dimensions of an attribute together, but not connected, in this table, the different aspects of each attribute can be acknowledged within the service it is aligned with. For an example of how this has been applied in pracice please refer to the East Trinity Case Study report.

Matrix table: This table can be filled in using the following:

n/a = either no thematic presence in data, or already counted elsewhere on matrix.

1 = low thematic prevalence,

2 = moderate

3 = high thematic prevalence.

	Knowledge	Recreation	Harvesting	Identity	Access	Social Capital	Community cohesion	Caring for Country
SEEA Cultural Service 1: Recreation-related services		n/a						
SEEA Cultural Service 2: Visual amenity services								
SEEA Cultural Service 3: Educational, scientific and research services								
SEEA Cultural Service 4: Spiritual, artistic and symbolic services								
SEEA Cultural Service 5: Other cultural								
Wellbeing Services								
Stewardships services								
SEEA Provisioning Service: wild animals, plants and other biomass provisioning services								

[289] Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW.

Table 13.8: **Cultural services account table based on SEEA (supply).** Totals in the far-right column reflect those areas with greater diversity of ecosystem types, afford a richer overall collection of cultural service. Data from Trinity Inlet case study[290].

		Selected ecosystem type					
	Available units of measure	**Mangrove**	**Saltmarsh**	**Supratidal forest**	**Water bodies**	**Other landcovers**	**Total**
SEEA Cultural Service 1: Recreation-related services	*Counts of distinct thematic types, ways or actions (e.g.fishing, camping, family gatherings)*	4	4	4	4	n/a	16
SEEA Cultural Service 2: Visual amenity services	*Counts of distinct thematic types, ways or actions (e.g. growth, recovery of habitat)*	2	2	3	3	1	11
SEEA Cultural Service 3: Educational, scientific and research services	*Counts of distinct thematic types, ways or actions (e.g. scientists collecting data, biological surveys, school visits)*	6	6	4	3	1	11
SEEA Cultural Service 4: Spiritual, artistic and symbolic services	*Counts of distinct thematic types, ways or actions (e.g. sites, dances, artworks)*	4	4	4	4	1	17
SEEA Cultural Service 5: Other cultural	*Counts of distinct thematic types, ways or actions*	2	1	1	1	n/a	5
Wellbeing Services	*Counts of distinct thematic types, ways or actions (e.g. employment)*	3	3	3	3	n/a	12
Stewardships services	*Counts of distinct thematic types, ways or actions (e.g. knowledge passed on)*	2	2	2	2	2	10
SEEA Provisioning Service: wild animals, plants and other biomass provisioning services	*Counts of distinct thematic types, ways or actions (e.g. hunting, medicine)*	6	3	16	n/a	3	28
SEEA Provisioning Service – Wild fish and other natural aquatic biomass provisioning services	*Counts of distinct thematic types, ways or actions (e.g. fishing, crabbing)*	n/a	n/a	n/a	4	n/a	4

[290] Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW.

Table 13.9: Cultural services account based on the SEEA (Use). Data from Trinity Inlet case study[291].

		Economic units	
	Available units of measure	**Households**	**Total**
SEEA Cultural Service 1: Recreation-related services	*Counts of distinct thematic types, ways or actions (e.g. fishing, camping, family gatherings)*	16	16
SEEA Cultural Service 2: Visual amenity services	*Counts of distinct thematic types, ways or actions (e.g. growth, recovery of habitat)*	11	11
SEEA Cultural Service 3: Educational, scientific and research services	*Counts of distinct thematic types, ways or actions (e.g. scientists collecting data, biological surveys, school visits)*	19	19
SEEA Cultural Service 4: Spiritual, artistic and symbolic services	*Counts of distinct thematic types, ways or actions (e.g. sites, dances, artworks)*	17	17
SEEA Cultural Service 5: Other cultural	*Counts of distinct thematic types, ways or actions*	5	5
Wellbeing Services	*Counts of distinct thematic types, ways or actions (e.g. employment)*	12	12
Stewardships services	*Counts of distinct thematic types, ways or actions (e.g. knowledge passed on)*	10	10
SEEA Provisioning Service: wild animals, plants and other biomass provisioning services	*Counts of distinct thematic types, ways or actions (e.g. hunting, medicine)*	28	28
SEEA Provisioning Service – Wild fish and other natural aquatic biomass provisioning services	*Counts of distinct thematic types, ways or actions (e.g. fishing, crabbing)*	4	4

291 Nursey-Bray, M., et al. (2023). Accounting for benefits from coastal restoration: a case study from East Trinity Inlet. Report to DCCEEW.

Key assumptions or limitations

The process of implementing and documenting a cultural account in a way that enables integration within the wider SEEA Framework, is a process needing combined high-level engagement and partnership with Indigenous peoples, as well as development of tables that can be used in tandem with the other account domains. Key insights provide additional understanding that support a best practice cultural accounting process.

i. The actual process of engagement, which is the first step in any cultural account needs to be undertaken with care and respect. Paying attention to who are the appropriate contacts, who can speak for Country and how the engagement/accounting will occur is the most important part of the process. A few factors will help facilitate this engagement.

   It is important to remember that wherever the proposed site may be, that in Australia ALL territory will be someone's Country: all of Australia is Indigenous Country. Hence when it appears that there is no active Indigenous presence, there will always be someone to contact that belongs to that Country (even if they do not live in the area), and the effort needs to be made to ensure that the appropriate Indigenous peoples have been identified and approached.

   Recommendation: That a cultural account of some kind is undertaken for any SEEA Accounting project.

ii. The development of genuine co-design and partnership opportunities will ensure the process and account is driven and underpinned by Indigenous cultural perspectives. Thus, in the first instance, establishing opportunities for the relevant Indigenous groups to lead or undertake the cultural account is important. However, if this is not possible, select consultants or researchers who either

have experience working with and/or have long standing relationships with the relevant Indigenous groups. Doing so will help facilitate timely but also open and transparent recording as well as enable establishment of productive co-partnerships with the Indigenous groups. Selection of the right person to work with Indigenous groups will also mean that there is a higher likelihood of obtaining deeper knowledge and detail and in a shorter period of time.

Recommendation: Provide co-leadership or partnership opportunities to and with the relevant Indigenous group/s.

iii. It is important to make sure that there is time to do the work, and to do site visits. Time pressures in any case study can be intense and are likely only to be surmounted by the willingness of all parties to work together to meet milestones. Yet relationship building with Indigenous peoples can be time consuming and time is needed to facilitate the trust required for deep and significant information exchange.

Recommendation: Make appropriate provision of time to do this kind of work.

iv. Indigenous peoples are, like everyone else, working. There is too often a subliminal expectation that because Indigenous peoples love their Country, they participate in processes like this 'for free'. Yet, they are being required to build partnerships, facilitate knowledge collection, as well as offer advice and guidance about its appropriate and effective use. All this work takes time and money. Deadlines for accounting deliverables are best met, if project action plans take into account that Indigenous people have other commitments. Timely deployment of resources to develop Indigenous collaborations, and maintain positive relationships, is critical.

Recommendation: That formal financial allocations are made to Indigenous team members, groups, or representatives as a recognition of their contribution to the accounting project.

v. Indigenous peoples are very culturally diverse. Acknowledgement of this cultural diversity needs to be an active principle, and hence, it should not be assumed that what worked in one cultural account, will work in another. There will be differences in location, language, history and legacy of colonization, patterns of migration and dispossession, cultural mores and laws and levels of knowledge. As such every engagement requires negotiation from the beginning in terms of how engagement and data collection will proceed. Doing so will ensure that all ethical obligations and requirements are undertaken. Co-drafting this process with Indigenous groups provides an opportunity to openly discuss each other's expectations.

Recommendation: As a first principle of engagement, acknowledge and find ways to embrace the cultural diversity and groups that may need to be involved; and who will either deliver or benefit from cultural and ecosystem services.

vi. An important and ongoing dynamic in the suggested cultural accounting process is a commitment to cultural review. This forms part of the co-design process. At multiple points there are opportunities to seek cultural review - of the methods proposed, the findings, write ups and the account tables. Reports can then be amended to include additional Indigenous insights and to correct any errors in how cultural information has been interpreted.

Recommendation: Embed a cultural review process as an essential part of development of a cultural account.

vii. Overall, for cultural accounts of Indigenous Australians, the difference between Indigenous Country and 'the site' as the focus of an account, must be acknowledged. Most restoration sites will be bounded territories, yet for the Traditional Owners it may be but one of many areas within their Country. So Indigenous views, perspectives, and experience of cultural and ecosystem services, will still be couched in the broader concept of Country. Future accounts could develop ways of acknowledging this, especially as some of the observed changes do not just accrue from the restoration itself, but external factors that affect the site. The physical bounding of a site in this way can restrict the gathering of potentially useful information. Values and services thus need to be articulated within a paradigm of connection and the tables in this chapter reflect this circularity, the fundamental holism that is embedded in the idea of Country, and the reciprocity it represents.

Recommendation: Acknowledge that the chosen site will be understood as part of Country, and not understood as an assemblage of parts to be individually considered (i.e., the different SEEA accounts).

viii. Further, Traditional Owners do not just benefit from ecosystem services – which are the focus of current SEEA models, but they also assert the prioritisation of the cultural importance of them being able to Care for Country – they undertake activities that benefit nature. This belief is because Indigenous peoples do not just own Country – it owns them, and healthy Country equals healthy people. Stewardship towards Country and people is inseparable from all layers of cultural value. The atomistic separation of services in a SEEA account does not align with cultural understanding of how ecosystem services work, where benefits are given as well as accrued.

Recommendation: Scope needs to be found within the SEEA accounting tables to incorporate levels of connection and it is suggested that that Traditional Owner connections offer an important contribution to accounting practice, given that nature is itself so inter-connected.

13.3 Conclusions

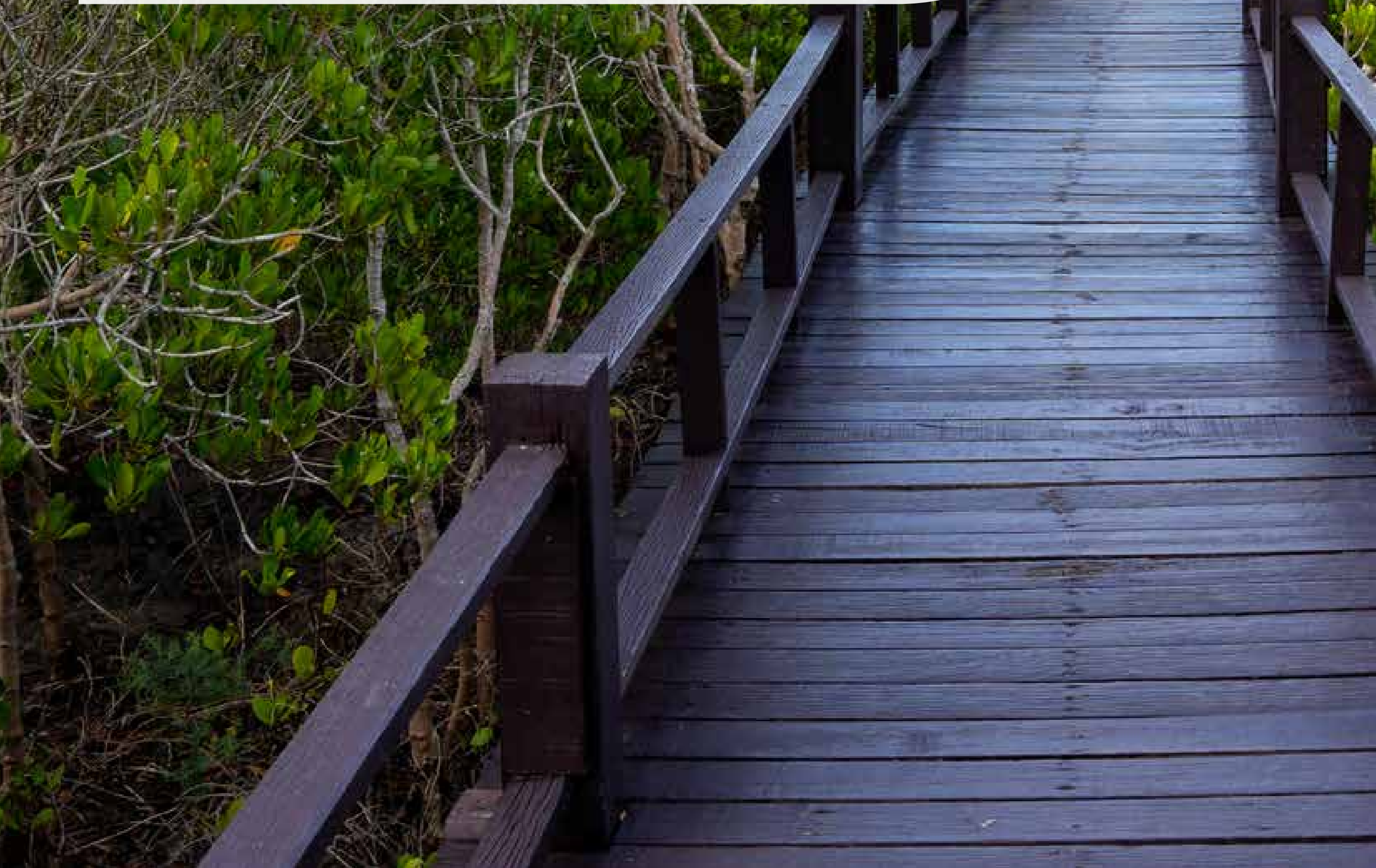
The development of a cultural account requires acknowledgment of cultural pluralism as well as a way of estimating the value of ecosystems services; giving them a holistic and circular foundation rather than the atomistic and linear modes that SEEA models have. Current SEEA models need to be extended to capture the plethora of values that occur in cultural accounts.

The process suggested in this chapter will result in production of a cultural account that can be used in conjunction with other SEEA accounts to provide an integrated and interdisciplinary understanding for purposes of Blue Carbon and other forms of accounting overall. The incorporation of Indigenous values and inherent circularity in approaches into current ecosystem service accounting, will enable engagement with different but very sophisticated, millennia old knowledge systems. This approach also helps to highlight and account for the ‘value adding’ that Indigenous resource management practices represent within environmental systems today.

Environmental protection accounts

Introduction to environmental protection accounts (Sections 14 and 15)

In addition to measuring changes on ecosystem extent, condition and ecosystem services, it is important to record data on the actual interventions themselves as part of the restoration project, both in physical terms (restoration actions) and monetary terms (restoration expenditures). This will allow for consideration of the most successful actions, as well as facilitate any analysis of the cost-effectiveness of investment over time. In this section, the guide considers restoration activities (physical accounts) and restoration expenditure (monetary accounts).



14. Restoration activities (physical accounts)

Prof. Will Glamore, Dr Alice Harrison

14.1 Summary of section

Effective restoration of physical ecosystems is a pathway towards improved ecosystem service provision. On this pathway, it is important to understand the drivers of change, which are recorded in the environmental activity accounts. The physical activity accounts record and quantify activities that have taken place for the purpose of environmental protection/restoration and allow tracking of resources and efforts required to successfully restore ecosystems.

Physical restoration activities in the context of blue carbon ecosystems can be varied, depending on the existing environment and target ecosystem (e.g. mangrove, saltmarsh, or seagrass). However, it is likely to include aspects of the following:

i. Decommissioning or modification of existing infrastructure, such as:
   a. Floodgates, weirs or other tidal barriers.
   b. Levees or bund walls.
   c. Artificial drainage channels.
   d. Fencing.
   e. Breakwaters.

ii. Commissioning of new infrastructure, such as:
   a. Floodgates, weirs or other tidal barriers.
   b. Levees or bund walls.
   c. Artificial drainage channels.
   d. Fencing.
   e. Breakwaters.

iii. Planting/seeding of target vegetation.

iv. Management or eradication of invasive pests or weeds.

v. Chemical treatment of soil or water.

vi. Labour (measured in number of days), including project and site management, and technical support.

Methods

Restoration of blue carbon ecosystems may take form via a number of different pathways, depending on the existing land use/environment and the target ecosystems (e.g. mangroves, saltmarsh, seagrass etc.). Mapping of the project area using GIS software, prior to and following restoration, is required to quantify the on-ground restoration activities.

Existing hydrological layers and high-resolution aerial imagery (see data sources) can be supplemented with on-ground site inspections utilising a handheld GPS unit (capable of accuracy of at least 5 m) to map existing infrastructure (e.g. drainage networks, floodgates, levees, or breakwaters). It is expected that all lengths and areas will be accurate within 10 m. Using the mapping created, physical activities should be tabulated, as shown in the supplementary material.

Other restoration activities that cannot be spatially mapped, such as pest animal management or labour, should be recorded directly into the table. It is not expected that all aspects of the table below will to be relevant to all projects, and rows may be deleted, if required. Additional rows may be added to reflect restoration activities that have not been included in this list.

Data sources

High resolution aerial imagery and existing hydrological layers will be useful in mapping the existing site, with some relevant examples of data sources provided in the supplementary material. However, site inspections will be required in most instances to map and quantify physical restoration accounts. Site investigations should be aided by a handheld GPS unit, capable of a minimum accuracy of 5 m, to allow for spatial mapping. Other surveying methods, such as GPS enabled drones or high accuracy RTK GPS systems may also be used if the technology is available. Annual data are needed for annual accounts.

In addition to mapping, records of days worked will be required for both:

- Greater confidence in primary data collected using preferred methods for valuation, relative to having to rely on values estimated using proxy methods such as benefit transfer.
- Technical support and consultation. This includes the time expended to complete tasks essential for the restoration, but not explicitly associated with on-ground activities. This may include modelling, design of restoration works, or expert advice. This is likely to include time from external consultants and stakeholders.

14.2 Detailed section

The restoration activities tabulate the effort required to achieve the restoration outcomes. Example restoration activities are shown conceptually in Figure 14.1. Restoration activities in the context of this guide includes activities that seek to increase the area (or improve the condition) of blue carbon ecosystems. Environmental activity accounts do not include activities whose primary purpose is to satisfy technical needs, safety, or security of an enterprise, regardless of whether there is an incidental environmental benefit.

Methods

The physical restoration activities should be tabulated as per Table 14.1, where rows not relevant to the project can be deleted as required. This should be accompanied by spatial mapping using GIS software of on-ground works, particularly including the project area, infrastructure commissioned/ modified/ decommissioned. At a minimum, mapping must identify:

- The project area.
- All drainage and/or protection infrastructure that is required to maintain the restored ecosystem. This includes infrastructure that is removed or modified in the restoration process. This will vary from site to site, depending on the existing and target environment, but may include drainage networks, tidal exclusion barriers, levees, breakwaters, and/or fencing.
- Areas where one-off activities are required, such as weeding, or chemical treatment of acid sulphate soils.

The cost of this tabulation and mapping is unlikely to require significant additional costs, as long as land managers are aware of the need for record keeping throughout the process of restoration.

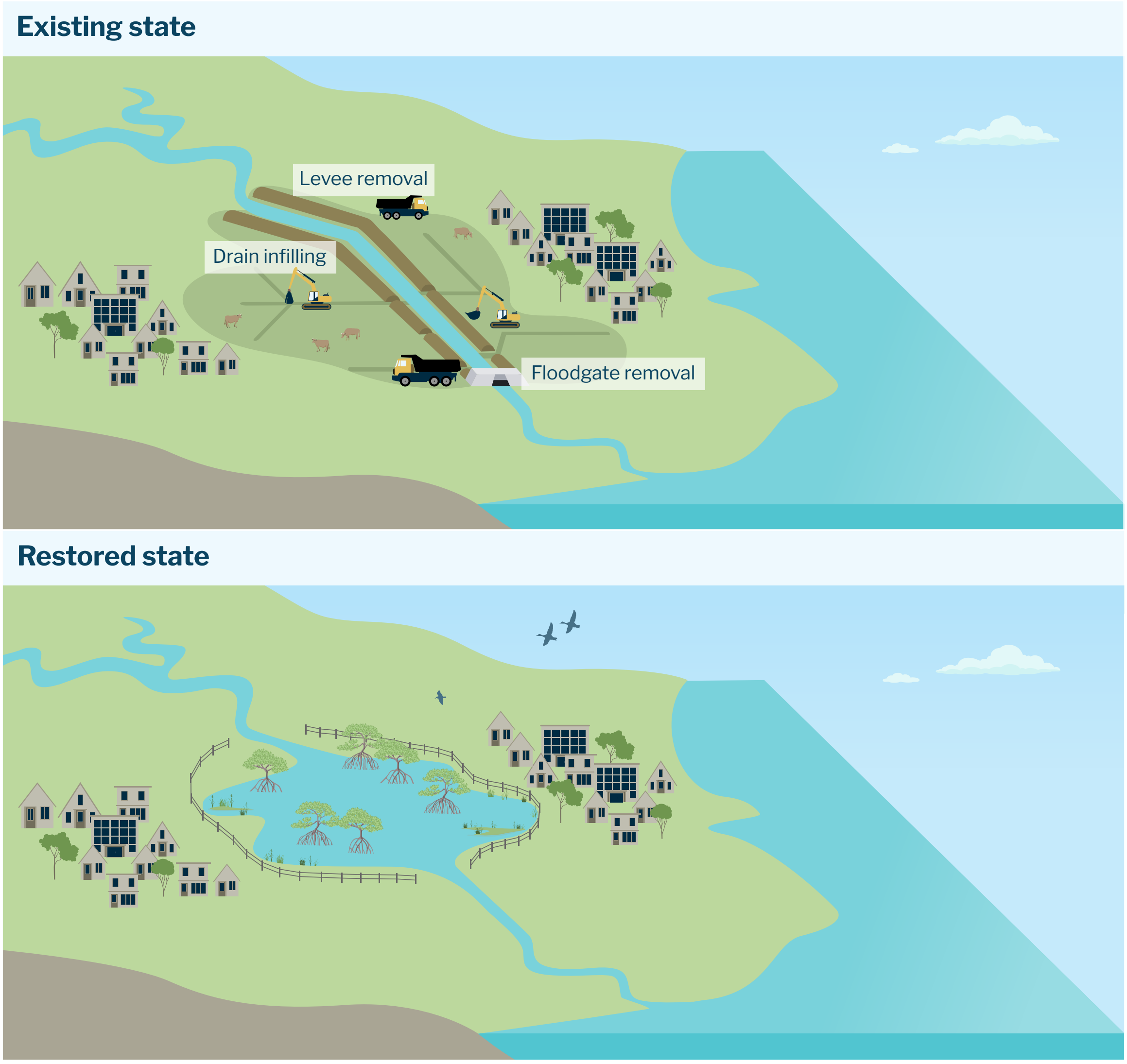


Figure 14.1: **Conceptual diagram of restoration activities (such as levee removal or drain infilling).**

Table 14.1: Restoration activities (physical).

Category	Item	Units	Comments
Physical landscape	Project area	Ha	This includes the total project area, and is not habitat specific (included in extent accounts)
Decomissioned/ modified infrastructure	Number of floodgates removed/modified	#	
	Number of weirs removed or modified	#	
	Length of levee/bund removed or modified	m	
	Length of drainage infilled or modified	m	
	Length of fencing removed	m	
	Length of breakwater removed	m	
Comissioned infrastructure	Number of floodgates/tidal exclusion structures commissioned	#	
	Length of levee/bund constructed	m	
	Length of drain contructed	m	
	Length of fencing installed	m	
	Length of breakwater constructed	m	Material to be included. May include submerged breakwaters, or sandbag structure
Planting	Number of plantings (or volume of seeds used)	# (or kg)	Species should be specified. Multiple rows may be required.
Pest and weed control	Pest animals eradicated	#	Species to be provided, where possible. Multiple rows to be used as required
	Area of weeds removed	m^2	Species and methods to be provided where possible. Multiple rows to be used as required.
Chemical	Volume of soil treated	m^3	e.g. liming of acid sulphate soils. Details of treatment to be provided.
	Volume of water treated	m^3	Details of treatment to be provided.
Labour	Project and site management	# days	To coordinate on-ground restoration works, site maintenance and any project approvals prior to restoration.
	Technical support	# days	Includes technical studies, such as modelling, design or consultants.

Data sources

A knowledge of on-ground remediation work is essential for the mapping and quantification of the physical restoration activities. Site specific data collection using a hand-held GPS (or alternative, more accurate methods) and records of work completed is expected to be required in most instances. However, there are a number of state-wide layers which may assist in mapping, tabulated in Table 14.2. Additional mapping resources may be used as required to complete the mapping of physical restoration activities.

Table 14.2: List of publicly accessible data sources that may be used in quantifying physical restoration accounts.

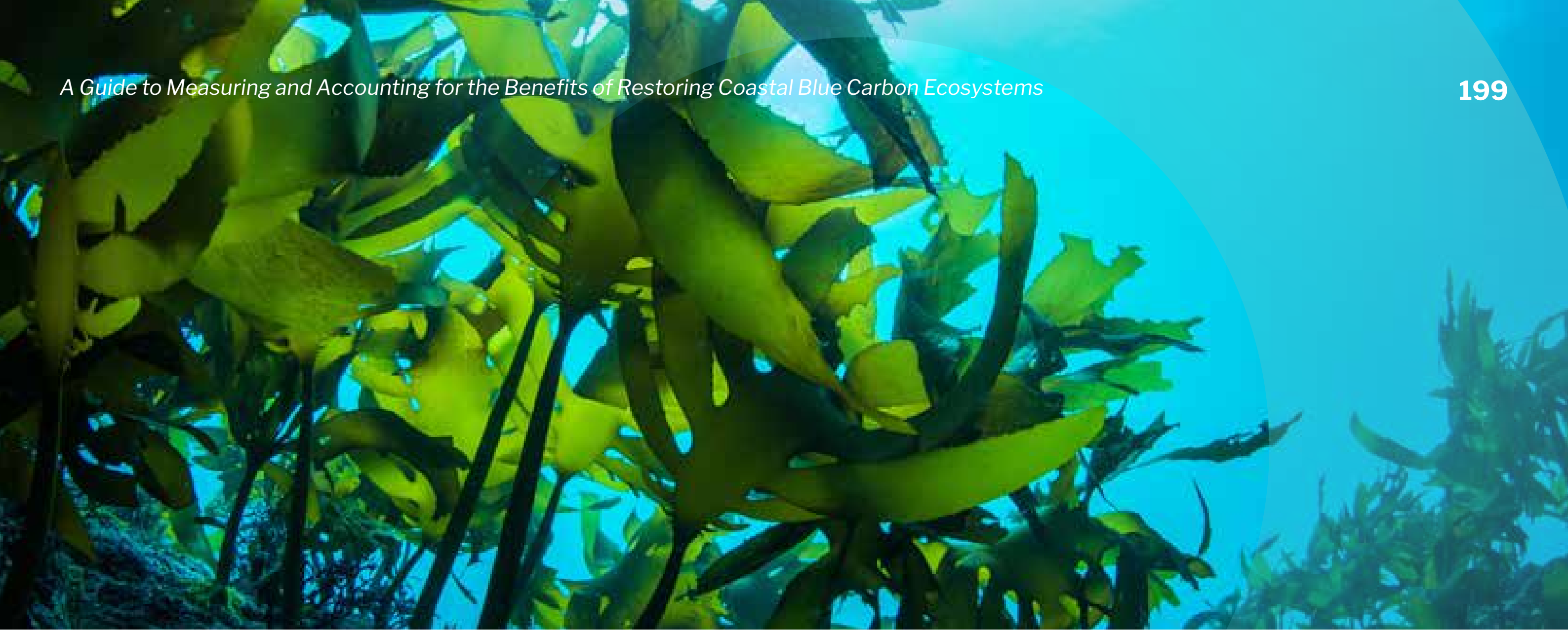
Resources	Coverage	Data available
Nearmap (https://www.nearmap.com/au/en)	AUS (coverage varied)	High resolution aerial imagery
Geoscience Australia National Surface Water Information (https://www.ga.gov.au/scientific-topics/national-location-information/national-surface-water-information)	AUS	Surface water hydrology layers
Google earth (https://earth.google.com/web/)	AUS	Medium resolution aerial imagery
Geoscience Australia Elevation Foundation Spatial Data (https://www.ga.gov.au/scientific-topics/national-location-information/digital-elevation-data)	AUS	Digital elevation models
NSW Spatial Collaboration Portal (https://portal.spatial.nsw.gov.au/portal/apps/sites/#/home)	NSW	High resolution aerial imagery Hydrological layers Digital elevation models
Queensland Globe (https://qldglobe.information.qld.gov.au/)	QLD	High resolution aerial imagery Digital elevation models
VicMap (https://mapshare.vic.gov.au/mapsharevic/) CoastKit (https://mapshare.vic.gov.au/coastkit/)	VIC	High resolution aerial imagery Hydrological layers
The List (https://maps.thelist.tas.gov.au/listmap/app/list/map)	TAS	High resolution aerial imagery Vegetation mapping
LocationSAMapViewer (https://location.sa.gov.au/viewer/)	SA	High resolution aerial imagery Hydrological layers Vegetation mapping

Table 14.2: cont.

Resources	Coverage	Data available
NatureMaps (https://data.environment.sa.gov.au/NatureMaps/Pages/default.aspx)	SA	Hydrological layers Vegetation mapping Bathymetry
Landgate (https://www0.landgate.wa.gov.au/maps-and-imagery)	WA	High resolution aerial imagery
NR Maps (https://nrmaps.nt.gov.au/water.html)	NT	High resolution aerial imagery Hydrological layers

Key assumptions or limitations

The physical restoration accounts will be used to quantify the effort required to achieve the desired outcomes. It is assumed that these accounts will be completed by (or with the assistance of) individuals who have a detailed knowledge and understanding of the physical works completed on the site. Some minor works, such as opportunistic weeding, may be difficult to capture in these accounts, as they aren't readily visible in aerial imagery, or do not require the purchase of specialised material (such as chemicals or seeds), which may be required to be accounted as labour for site management (in days worked). Days attributed to project or site management labour will require documentation of days worked by site managers. Similarly, days attributed to technical support should also be sourced from records where possible. However, as this will likely include the contributions of external consultants, estimates of days worked can be estimated from total contract costs and hourly staff rates if no further documentation is available.



15. Restoration activities (monetary accounts)

Dr Abbie Rogers, A/Prof Michael Burton, Dr Tafesse Estifanos, Dr Fitalew Taye

15.1 Summary of section

In line with the earlier discussion about different modes of valuing environmental assets, consideration of restoration costs can take two perspectives.

Within the SEEA framework, one can adopt a restoration cost-based approach to measuring degradation of an environmental asset, i.e. the cost of restoring the environmental asset to a prior state is a measure of the loss of value associated with that degradation. It should be noted that this is entirely a forward-looking perspective: it is an evaluation of the costs of restoring an existing degradation in the environmental asset to some pre-defined level, or an expected expenditure to avoid a future degradation. If the restoration has already happened, then there is no change in condition to evaluate and the restoration costs involved are not relevant, unless the accounts are being used for scenario analysis. An issue then is in predicting future quantities of inputs and prices that might be associated with those restoration activities. It should be noted that this approach to estimating the cost of degradation may bear little or no relationship to the value of the asset as estimated through some measure of exchange value (simulated or otherwise) i.e. the degradation of a marine ecosystem may be extremely difficult and costly to restore but it may provide little in terms of societal ecosystem services as conventionally measured within the SEEA-EA framework.

Restoration costs are more easily placed within the framework of welfare values and costs benefit analysis: in fact, they are central to any evaluation of whether a restoration activity generates net benefits to society. Given an evaluation of the surplus measures of welfare that have been evaluated as having been lost through some change in environmental condition (or potentially gained through some improvement), these can be compared to the economic costs of restoring the ecosystem, and judgements made on whether benefits exceed costs. These may either be prospective, if an evaluation is being made for a future restoration, or historical if an ex-post evaluation is being made of a project. The outcome of the latter obviously cannot change what actions have been already undertaken, but they may be used to inform future decisions, or consider the processes by which previous decisions were made.

In terms of measures of economic activity, restoration costs may give some indication of the contribution of the restoration activity to the 'restoration economy'[292] in terms of temporary jobs or regional economic activity to implement the change, but strictly these are costs associated with repairing environmental damage rather than benefits.

15.2 Detailed section

Methods

To some extent, the methods associated with estimating restoration costs are straightforward: having identified the physical inputs (outlined above) estimates of the market value of those inputs is required, following simple accounting processes. An issue arises with the intertemporal nature of restoration. It is unlikely that ecological restoration will occur immediately following the expenditure, and it is likely that to achieve a degree of restoration costs will need to be incurred over a number of years. Within the SEEA framework, the potential for future costs represents a liability for future provision, which is not something that normally occurs within the national accounting framework.

Within the welfare analysis framework, methods to deal with cost occurring across time are well established: costs need to be brought to a common year as a baseline, through applying a discounting framework, which are then compared with benefits, also brought to the same year. A distinction has to be made between capital costs (that occur once) and variable or recuring costs which may be assumed to occur throughout the period of the restoration project and potentially beyond if ongoing maintenance is required.

Data sources

A key consideration in the evaluation of costs of restoration relates to capital v. recurring costs, and the treatment of costs across time. Central to this is the concept of discounting, which brings costs incurred at different points in time to a common metric (i.e., net present value). Selection of the rate used may depend on the proponent: if costs are being incurred by public bodies, then national discount rates should be applied, while private proponents may see their private discount rate as more appropriate.

Key assumptions or limitations

In some cases, restoration activities are undertaken by volunteers. The treatment of this input into the process could have significant consequences for the overall costs of the project. Current thinking suggests that volunteer activity should be considered as having a net benefit to individuals, and hence there should be no inferred cost of volunteer time included in the assessment of costs when estimating restoration costs for welfare analysis.

A decision has to be made on the appropriate discount rate to apply. In the case of public sector investment in infrastructure, the current recommendation is that a 7 % discount rate should be used[293] as a measure of the social discount rate, with sensitivity analysis conducted at 4 and 10 %. Private sector investors may feel it more appropriate to use the market rate of interest relevant to their investments.

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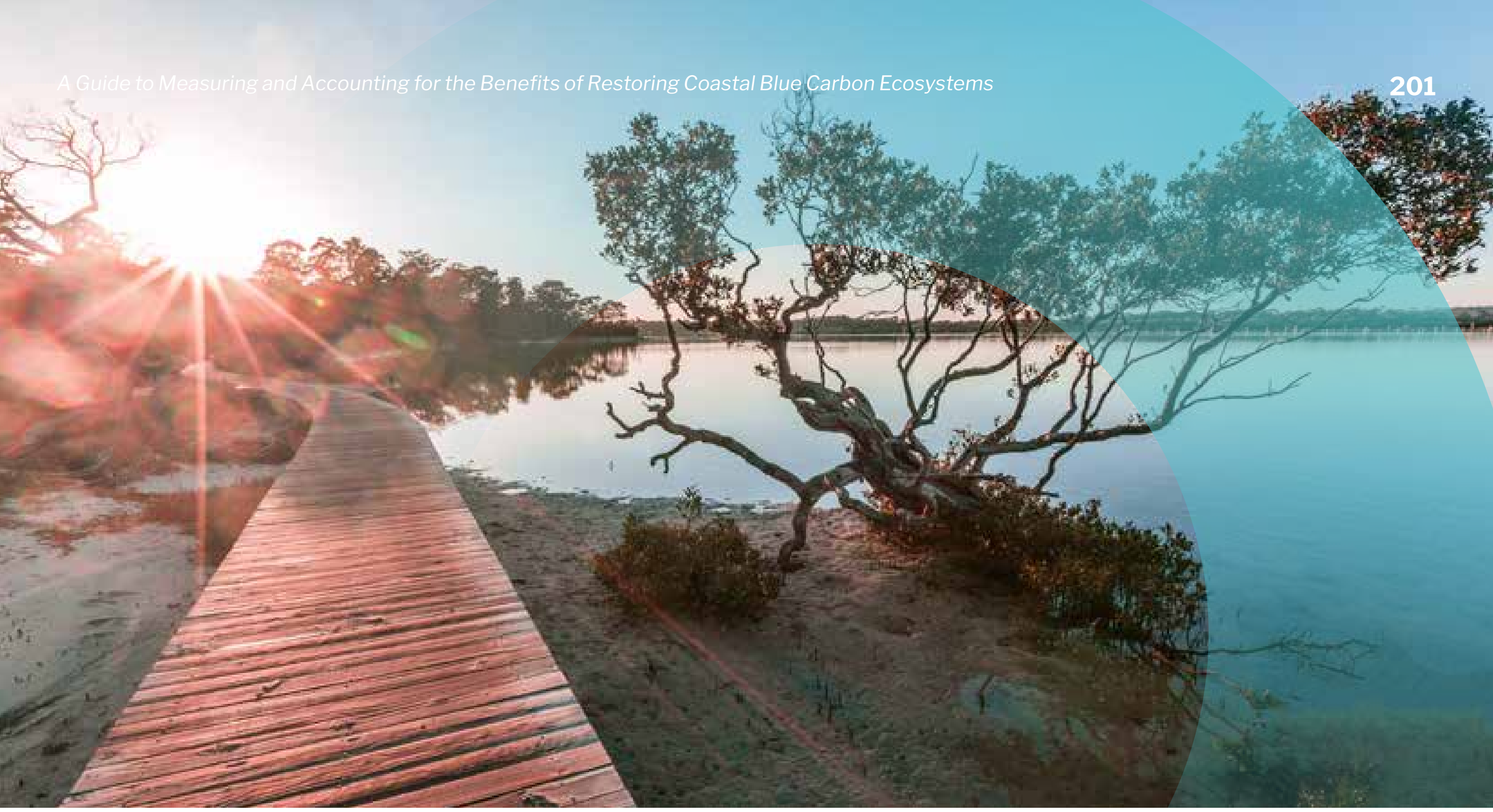
[292] e.g. BenDor et al. 2015; https://doi.org/10.1371/journal.pone.0128339

[293] Infrastructure Australia. (2021) Guide to Economic Appraisal https://www.infrastructureaustralia.gov.au/sites/default/files/2021-07/Assessment%20Framework%202021%20Guide%20to%20economic%20appraisal.pdf



16. EEA outputs and presentation

16.1 Introduction

In previous sections, the guide has provided detailed methodologies and data sources which can be used to assess various impacts of a restoration project in a coastal blue carbon ecosystem. Below, the guide provides examples of tables that can be used to present data outputs in a format aligned with the SEEA-EA. Tables cover ecosystem extent, condition, physical ecosystem services, monetary ecosystem services, and restoration accounts.

Tables have been prepared by the project team to present data in the two site-level assessments and are not official SEEA-EA tables.

16.2 Ecosystem extent account

Table 16.1: **An example table that would be used to show change in different ecosystem types before and after restoration actions have taken place.** Supratidal swamp forests technical are classified within the same category as mangroves (Intertidal forests and shrublands MFT1.2) but have been split here. Re-evaluation is changes due to, for example, use of data at different resolution or imputation of cells with cloud cover.

Realm	**Marine-Freshwater-Terrestrial**			**Marine**		**Marine-Terrestrial**		**Terrestrial**	**Freshwater**	
Biome	**MFT1 Brackish tidal**			**M1 Marine shelf**		**MT1 Shorelines biome**		**T7 Intensive land use**	**F3 Artifical wetlands**	
Selected Ecosystem Functional Group (EFG)	Supratidal swamp forest MFT1.2* ha	Saltmarsh MFT1.3 ha	Mangroves MFT1.2 ha	Seagrass MFT1.1 ha	Subtidal sand beds MFT1.7 ha	Muddy shorelines MFT1.2 ha	Sandy shorelines MFT1.3 ha	Derived semi-natural pastures T7.5 ha	Constructed lacustrine wetlands F3.2 ha	**Total ecosystem extent** ha
Opening extent (pre-restoration)										
Additions										
Managed expansion										
Unmanaged expansion										
Total additions										
Reductions										
Managed reduction										
Unmanaged reduction										
Total reductions										
Net change										
Closing extent										

16.3 Ecosystem condition account

Table 16.2: **Example ecosystem condition indicator account table. Values for connectivity of ecosystem are mean of all cells in restoration activity boundary.** Comparison area for opening and closing mean values is the overlapping areas of the same ecosystem type (i.e. where mangrove was present in both pre- and post-restoration activities. Note that for vegetation cover, biomass, greenness and wetness this is reported as change in hectare area for descriptor (i.e. opening value = area gained or maintained in value of descriptor, closing value = area loss in value of descriptor, change in indicator = net change in area for condition indicator). Continued over page.

Ecosystem type	SEEA Ecosystem Condition Typology Class			Variable values		Reference level values		
		Descriptor	Measurement unit	Opening value	Closing value	Upper level (e.g. Natural)	Lower level (e.g. Collapse)	Change in indicator
Mangrove	Physical state	Landscape wetness Spectral index						
	Chemical state	Salinity	ppt					
	Chemical state	pH	pH scale					
	Compositional state	Fish diversity	Shannon's index					
	Structural state	Mangrove cover Above-ground biomass	% cover Mg ha^{-1}					
	Functional state	Vegetation greenness	Spectral index					
	Seascape characteristics	Connectivity of ecosystem	Index					
Saltmarsh	Physical state	Landscape wetness	Spectral index					
	Chemical state	Salinity	ppt					
	Chemical state	pH	pH scale					

Table 16.2: cont.

Ecosystem type	SEEA Ecosystem Condition Typology Class	Descriptor	Measurement unit	Variable values		Reference level values		
				Opening value	Closing value	Upper level (e.g. Natural)	Lower level (e.g. Collapse)	Change in indicator
Saltmarsh	Compositional state	Bird community diversity	Shannon's index					
	Structural state	Saltmarsh cover Above-ground biomass	% cover Mg ha^{-1}					
	Functional state	Vegetation greenness	Spectral index					
	Seascape characteristics	Connectivity of ecosystem	Index					
Supratidal forests	Chemical state	Salinity	ppt					
	Chemical state	pH	pH scale					
	Compositional state	Age since restoration activities	Years					
	Structural state	Vegetation cover Above-ground biomass	% cover Mg ha^{-1}					
	Functional state	Vegetation greenness	Spectral index					
	Seascape characteristics	Connectivity of ecosystem	Index					
Muddy shorelines	Physical state	Landscape wetness	Spectral index					
	Chemical state	pH	pH scale					
	Compositional state	macroinvertebrate community diversity	Shannon's index					

Table 16.2: cont.

Ecosystem type	SEEA Ecosystem Condition Typology Class	Descriptor	Measurement unit	Variable values		Reference level values		
				Opening value	Closing value	Upper level (e.g. Natural)	Lower level (e.g. Collapse)	Change in indicator
Muddy shorelines	Compositional state	Bird community diversity	Shannon's index					
	Structural state	Vegetation cover	% cover					
	Functional state	Vegetation greenness	Spectral index					
	Seascape characteristics	Connectivity of ecosystem	Index					
Derived semi-natural pastures	Physical state	Landscape wetness	Spectral index					
	Chemical state	pH	pH scale					
	Compositional state	Bird community diversity	Shannon's index					
	Structural state	Native vegetation cover	% cover					
	Functional state	Vegetation greenness	Spectral index					
	Landscape characteristics	Connectivity of ecosystem	Index					

16.4 Physical ecosystem service account

Table 16.3: Example ecosystem services supply and use account table in physical terms.

							Marine-Freshwater-Terrestrial			Marine		Marine-Terrestrial		Terrestrial	Freshwater	Total
		Industry				Sector	MFT1 Brackish tidal			M1 Marine shelf		MT1 Shorelines biome		T7 Intensive land use	F3 Artifical wetlands	
Supply	Units of measure	Agriculture, forestry and fishing	Other industry	Total industry	Households	Government	Supratidal swamp forest	Saltmarsh	Mangroves	Seagrass	Subtidal sand beds	Muddy shorelines	Sandy shorelines	Derived semi-natural pastures	Constructed lacustrine wetlands	
							MFT1.2*	MFT1.3	MFT1.2	M1.1	M1.7	MT1.2	MT1.3	T7.5	F3.2	
Provisioning																
Wild fish and other natural aquatic: School Prawn	kg															
Total biomass	kg															
Regulating and maintenance services																
Nursery population and habitat maintenance services	kg ha^{-1} y^{-1}															
Cultural services Recreation-related services																
Recreational fishing	No. of fishing days															
Bird watching	No. of trips															

16.5 Monetary ecosystem service account

Table 16.4: **Example Ecosystem services supply account in monetary terms.**

							Marine-Freshwater-Terrestrial			Marine		Marine-Terrestrial		Terrestrial	Freshwater	
		Industry			Sector		MFT1 Brackish tidal			M1 Marine shelf		MT1 Shorelines biome		T7 Intensive land use	F3 Artifical wetlands	Total
Supply	Units of measure	Agriculture, forestry and fishing	Other industry	Total industry	Households	Government	Supratidal swamp forest	Saltmarsh	Mangroves	Seagrass	Subtidal sand beds	Muddy shorelines	Sandy shorelines	Other land covers	Constructed lacustrine wetlands	
							MFT1.2*	MFT1.3	MFT1.2	M1.1	M1.7	MT1.2	MT1.3	T7.1	F3.2	
Provisioning																
Wild fish and other natural aquatic																
Blue Swimmer	AUD$															
Crabs																
Dusky Flathead	AUD$															
Giant Mud Crab	AUD$															
Luderick	AUD$															
Total biomass	AUD$															
Regulating and maintenance services																
Water quality (Nitrogen)	AUD$															
Cultural services Recreation-related services																
Recreational fishing	AUD$															
Bird watching	AUD$															

16.6 Restoration account

Table 16.5: **Example monetary account table for restoration activities.**

Year	Mangroves	Saltmarsh	Seagrass	Supratidal	Others	Total annual costs unadjusted (AUD)	CPI adjusted total annual cost (AUD)
2006							
2007							
2008*							
2009*							
2010							
2011							
2012							
2013							
2014							
2015							
2016							
2017							
2018							
2019							
2020							
2021							
2022							
Total cost (AUD)							
Post-restoration							

16.7 Regulation and maintenance

Table 16.6: **Example carbon asset account table.**

Realm	**Marine-Freshwater-Terrestrial**			**Marine**		**Marine-Terrestrial**		**Terrestrial**	**Freshwater**	
Biome	**MFT1 Brackish tidal**			**M1 Marine shelf**		**MT1 Shorelines biome**		**T7 Intensive land use**	**F3 Artifical wetlands**	
Selected Ecosystem Functional Group (EFG)	Supratidal swamp forest MFT1.2* tonnes C	Saltmarsh MFT1.3 tonnes C	Mangroves MFT1.2 tonnes C	Seagrass MFT1.1 tonnes C	Subtidal sand beds MFT1.7 tonnes C	Muddy shorelines MFT1.2 tonnes C	Sandy shorelines MFT1.3 tonnes C	Other land covers T7.1 tonnes C	Constructed lacustrine wetlands F3.2 tonnes C	**Total ecosystem extent** ha
Opening extent 2006 (pre-restoration)										
Additions										
Managed expansion										
Unmanaged expansion										
Total additions										
Reductions										
Managed reduction										
Unmanaged reduction										
Total reductions										
Net change										
Closing extent										

Table 16.7: Example ecosystem services supply and use account table in physical terms.

							Marine-Freshwater-Terrestrial			Marine		Marine-Terrestrial		Terrestrial	Freshwater	
		Industry			Sector		MFT1 Brackish tidal			M1 Marine shelf		MT1 Shorelines biome		T7 Intensive land use	F3 Artifical wetlands	
	Units of measure	Agriculture, forestry and fishing	Other industry	Total industry	Households	Government	Supratidal swamp forest	Saltmarsh	Mangroves	Seagrass	Subtidal sand beds	Muddy shorelines	Sandy shorelines	Other land covers	Constructed lacustrine wetlands	Total
SUPPLY							MFT1.2*	MFT1.3	MFT1.2	M1.1	M1.7	MT1.2	MT1.3	T7.1	F3.2	
Regulating and maintenance																
Global climate regulation services																
Sequestration	tonnes CO_2eyr^{1}															
Storage	tonnes CO_2e															
USE																
Regulating and maintenance																
Global climate regulation services																
Sequestration	tonnes CO_2eyr^{1}															
Storage	tonnes CO_2e															

Table 16.8: Example flood mitigation supply and use account table in physical terms – supply and use table, post-restoration.

							Marine-Freshwater-Terrestrial			Marine		Marine-Terrestrial		Terrestrial	Freshwater	Total
		Industry			Sector		MFT1 Brackish tidal			M1 Marine shelf		MT1 Shorelines biome		T7 Intensive land use	F3 Artifical wetlands	
	Units of measure	Agriculture, forestry and fishing	Other industry	Total industry	Households	Government	Supratidal swamp forest	Saltmarsh	Mangroves	Seagrass	Subtidal sand beds	Muddy shorelines	Sandy shorelines	Other land covers	Constructed lacustrine wetlands	
SUPPLY							MFT1.2*	MFT1.3	MFT1.2	M1.1	M1.7	MT1.2	MT1.3	T7.1	F3.2	
Ecosystem services																
Flood mitigation	Number of properties protected															
USE																
Ecosystem services																
Flood mitigation	Number of properties protected															

Appendix 1: Additional resources

Dr Abbie Rogers, A/Prof Michael Burton, Dr Tafesse Estifanos, Dr Fitalew Taye

Data transformation

For ecosystem accounting, data needs to be spatially explicit and the data from each source needs to be aligned to the same spatial units – basic statistical units, ecosystem types and ecosystem accounting area. As part of this the spatial data infrastructure is essential.

Determining the transformations required for spatial data requires the categorisation of different data sets according to their spatial characteristics. These characteristics include coverage, type of spatial data (i.e. raster or vector) and spatial resolution. Spatial coverage can be full, partial or no coverage for any given area.

Data that has full spatial coverage does not require transformation. Imputation (e.g. via interpolation or use of average value) may be required when there is partial coverage. When a data set has no coverage of an accounting area, then there may be suitable methods (e.g. extrapolation) for generating data for the accounting areas.

The type of spatial data also affects the type of transformation required. Comparatively little effort is needed for raster (grid) data, although the resolution of the raster data affects the transformation required. Aggregation (upscaling) is required when data is finer resolution than the accounting area, while disaggregation (down-scaling) is required if the data is broader than the accounting area. Raster data from different data sets may be up or down scaled within accounting areas to graphically show the variations within accounting area at the same level of resolution.

Vector (polygons) data can be problematic, and its usefulness depends to what extent the vector data can be mapped into the accounting area. If it is entirely within the accounting area, then it may be used for this area. A method to create information for the area that falls outside the polygon but within the accounting area will be needed (e.g. assume the data applies evenly throughout the polygon). If the vector data partially overlaps the accounting area, then again, a method is needed to attribute the data from the polygon to the entire accounting area (e.g. assume that the data is evenly spread and hence a simple percentage overlap can be applied). In both cases, the modifiable areal unit problem, to which there is no real solution, arises as data is seldom evenly spread over a polygon) and must be dealt with as best as possible. For example, through the use of supplementary raster data to allocate the appropriate characteristics to the accounting area from the vector data. Point data, a type of vector data, needs to be interpolated or extrapolated to produce estimates for an accounting area.

Temporal transformation is the alignment of different data sets to accounting periods. Controlling for seasonality is particularly important for data sets collected in different months across different years. For example, the condition of wetland may be different in summer months compared to winter months due to varied levels of rainfall and runoff. To create a time series information from different years may be used to impute data for the missing years (e.g. through linear interpolation). Alignment of calendar years

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[294] UN et al. (2014, 2018 and 2021); https://seea.un.org/content/seea-central-framework https://seea.un.org/sites/seea.un.org/files/documents/EA/seea_ea_white_cover_final.pdf

to fiscal years is a common problem. It is usually easier to align biophysical data to the fiscal years used in economic statistics, than it is to align fiscal with calendar years.

Alignment with accounting concepts principles is the process of ensuring that data complies with accounting definitions, classifications, and standards. These are defined in the United Nations' System of Environmental-Economic Accounting (SEEA) . This ensures that accounts are comparable over time, can be scaled up, and are coherent with accounts for other areas and themes (e.g. water, land).

Alignment of data with accounting principles also ensures coherence between different types of accounts. In this the asset accounts, need to be consistent with the flow accounts.

Spatial units of observation and their aggregation

In ecosystem accounting the unit of observation is an area known as a basic statistical unit. Units of similar attributes are grouped into ecosystem asset types that may supply ecosystem services within a particular ecosystem accounting area. For proponents of a coastal blue carbon ecosystem restoration project, the ecosystem accounting area is the physical area affected by the restoration project. Figure A1.1 shows this for a hypothetical map grouped to ecosystem types within an ecosystem accounting area.

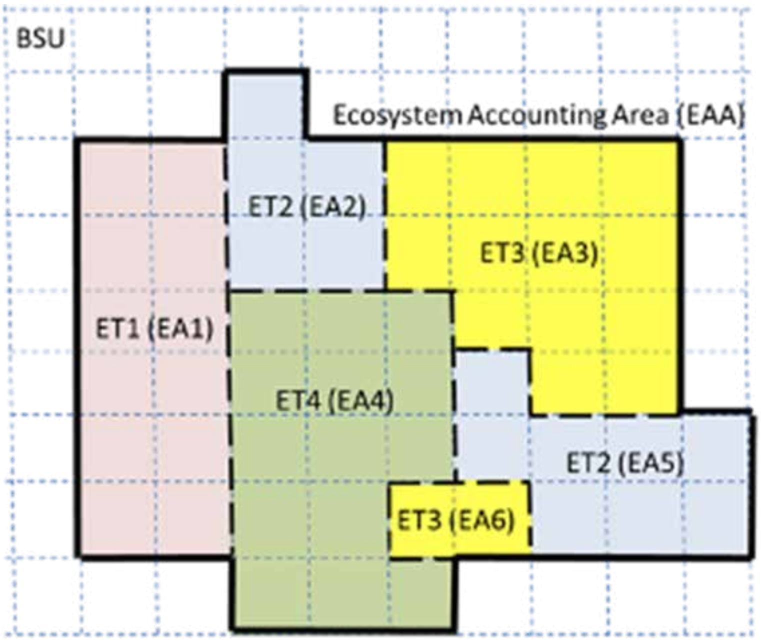

Figure A1.1: Relationship of units of observation (basic statistical units) to aggregations of ecosystem type with ecosystem accounting areas[295].

[294] UN et al. (2014, 2018 and 2021); https://seea.un.org/content/seea-central-framework https://seea.un.org/sites/seea.un.org/files/documents/EA/seea_ea_white_cover_final.pdf

[295] Source UN (2018). SEEA-EEA Tech Recs, Table 3.1 p. 40). Note that Ecosystem Assets (EA) represent individual, contiguous ecosystems. Ecosystem Types (ET) are EA of the same type.

Data quality assessment

Data quality frameworks are available from a range of international or national statistical agencies. For example, the Australian Bureau of Statistics[296], Eurostat[297], IMF[298], OECD[299] and Statistics Canada[300]. These are all similar and in general describe six dimensions of data quality and are:

1. **Relevance** – how well the statistics meets the needs of users in terms of the concept(s) measured, and the population(s) represented;
2. **Accuracy** – refers to the degree to which the data correctly describe the phenomenon they were designed to measure;
3. **Timeliness** – which is the delay between the reference period (the time to which the data pertain) and the date at which the data be-come available; and the delay between the advertised date and the date at which the data become available (i.e. the actual release date);
4. **Accessibility** – the ease of access to data by users, including the ease with which the existence of information can be ascertained, as well as the suitability of the form or medium through which information can be accessed;
5. **Interpretability** – the availability of information to help provide in- sight into the data;
6. **Coherence** – is the internal consistency of a statistical collection, product or release, as well as its comparability with other sources of information, within a broad analytical framework and over time.

These dimensions also reflect academic notions of data quality[301].

It is important to recognise that for decision making, data need to be more than accurate and there are often trade-offs between the various aspects of quality. Making information available when it is needed may require, for example, that timeliness be prioritised at the expense of accuracy.

While some aspects of data quality can be assessed objectively (these are quantifiable errors mentioned in the IPCC definition of un-certainty), an assessment of the wider concept of fitness-for-purpose is largely qualitative as it also brings to account other factors including user views, the soundness of methodologic practices and corporate culture within the agency compiling data.

Underlying the six dimensions of data quality is the notion of integrity – that information policies and practices are guided by ethical standards and professional principles which are transparent. The integrity of data producing agencies may be aided by the laws under which the agency operates and its willingness to subject its operations and performance to both internal and external scrutiny.

A key feature contributing to the integrity of national and environmental accounts is that they are usually released according to a predetermined schedule and the results are released to the public at virtually the same time as they released to the elected officials and government agencies (e.g. the Ministers for Finance or the Environment). This

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[296] ABS (Australian Bureau of Statistics), 2009. Data Quality Framework. ABS cat. no 1520.0., Canberra (Accessed 18 January 2018). http://www.abs.gov.au/ausstats/abs@.nsf/lookup/1520.0Main+Features1May+2009.

[297] Eurostat, 2005. Statistics Code of Practice. Eurostat, Luxemburg (Accessed 7 February 2018). http://www.dzs.hr/Eng/international/code_of_practice_en.pdf.

[298] IMF (International Monetary Fund), 2012. Data Quality Assessment Framework – Generic Framework. IMP, Washington, D.C (Accessed 12 February 2018). http://dsbb.imf. org/Pages/DQRS/DQAF.aspx.

[299] OECD (Organisation for Economic Co-operation and Development), 2012. Quality Framework and Guidelines for OECED Statistical Activities. OECD, Paris (Accessed 12 February 2018). http://www.oecd.org/officialdocuments/displaydocumentpdf/?cote=std/qfs(2011)1&doclanguage=en.

[300] Statistics Canada, 2002. Quality Assurance Framework. Statistics Canada, Ottawa (Accessed 12 February 2018). http://www5.statcan.gc.ca/access_acces/archive.action?loc=/pub/12-586-x/12-586-x2002001-eng.pdf.

[301] E.g. Clarke et al., (2011). Availability, accessibility, quality and comparability of monitoring data for European forest for use in air pollution and climate change science. iForest-Biogeosciences and Forestry 4(4),162-166, https://iforest.sisef.org/contents/?id=ifor0582-004

helps to ensure that the results are not altered or delayed by political or bureaucratic imperatives. Similar practices for ecosystem accounting would help ensure integrity.

SEEA key concepts and definitions

The SNA[302] and SEEA[303] contain many concepts and definitions. The main concepts and definitions needed for a basic understanding of ecosystem accounting are below, with sources provided for additional research.

SNA

1. **Asset**: An asset is a store of value representing a benefit or series of benefits accruing to the economic owner by holding or using the entity over a period of time. It is a means of carrying forward value from one accounting period to another (SNA para, 3.5). Assets are divided in classes (e.g. financial, produced and non-produced).
2. **Institutional unit**: An institutional unit is an economic entity that is capable, in its own right, of owning assets, incurring liabilities and engaging in economic activities and in transactions with other entities (SNA, para 4.2). Institution units are classified to sectors and industry based on their characteristics.
3. **Industry**: an industry consists of a group of establishments engaged in the same, or similar, kinds of activity (SNA, para 5.2).

Institutional units are the economic units of observation. Characteristics of economic units that are measured include, income, expenditure, employment, holdings or assets and liabilities.

SEEA-CF System of Environmental Economic Accounting – Central Framework

- Environmental assets
- Natural resources
- Products
- Residuals

SEEA-EA

- **Ecosystem condition** is the quality of an ecosystem measured in terms of its abiotic and biotic characteristics (SEEA-EA, para. 2.13)
- **Ecosystem extent** is the size of an ecosystem asset (SEEA-EA, para. 2.13)
- **Ecosystem services** are the contributions of ecosystems to the benefits that are used in economic and other human activity (SEEA-EA, para. 2.14)
- **Ecosystem type** (ET) reflects a distinct set of abiotic and biotic components and their interactions (para. 2.11).
- **Ecosystem conversions** refer to situations in which, for a given location, there is a change in ecosystem type involving a distinct and persistent change in the ecological structure, composition and function which, in turn, is reflected in the supply of a different set of ecosystem services (para. 4.23).

[302] SNA: System of National Accounts: https://unstats.un.org/unsd/nationalaccount/sna.asp

[303] SEEA: System Of Environmental Economic Accounting: https://seea.un.org/content/seea-central-framework

Classifications used in accounting

Many of the key concepts have associated classifications. Key classifications in the SNA and SEEA include:

1. Industries based on productive activity, as per the Australia New Zealand Standard Industry Classification[304]. E.g. agriculture, mining, manufacturing, health education, etc.
2. Sector based on their principal functions, behaviour, and objectives. These are households, not-for-profit institutions, government and financial and non-financial corporations (SNA para 2.17).
3. Goods and services (as defined in the Central Product Classification[305])
4. Ecosystem types
5. Ecosystem services

The SEEA-EA includes a reference classification of ecosystem types[306], while recognising that other classifications of ecosystem service types exist and may be appropriate in particular circumstances.

[304] https://www.abs.gov.au/anzsic

[305] https://unstats.un.org/unsd/classifications/Econ/cpc

[306] in Table 3.2 UN et al. 2021, p. 57 based on the Global Ecosystem Typology (Keith et al., 2020) and ecosystem services in Table 6.3 UN et al. 2021, pp. 131-134; https://seea.un.org/sites/seea.un.org/files/documents/EA/seea_ea_white_cover_final.pdf

Appendix 2: other economic tools for ecosystem service valuation

Economic valuation

The purpose of this guide is to set out the steps to employ SEEA-EA for blue carbon ecosystems. However, there are a range of economic tools available to assist with environmental decision making, each having their own strengths.

Economic valuation tools are different to economic accounting tools, in that they are used to understand the contribution that a project could make (or has made, if retrospective) to the welfare or wellbeing of society. Economic valuation uses 'welfare values' to measure the benefits or contribution that a restoration project makes. Welfare values are also quantified in monetary terms but capture the 'total economic value' of the project in terms of producer and consumer surplus; that is, a measure of total value generated to all stakeholders of a project, not a measure of value exchanged (which may well be less total value for restoration projects).

The total economic value directly includes things like non-use existence values. Given that economic valuation uses welfare values, there is no need to separate out non-use from use values as is the case in SEEA-EA monetary accounts with exchange values not being applicable to non-use and welfare values captured separately to the primary accounts in bridging tables.

Economic valuation includes the use of tools like benefit-cost analysis, where the benefits (i.e. the welfare values) of a restoration project can be compared against its costs to inform decision makers about whether a project should proceed (if benefits exceed costs), or to select the best of several alternative projects.

Which tool for what decision?

SEEA-EA and economic valuation (and benefit-cost analysis or BCA) each have different strengths. EEA offers a coherent framework for data organization, which if adopted at scale can create a common global dialogue for discussions and comparisons across nations and projects. Its ability to monitor trends and performance over time provides an indicator and opportunity to highlight when sustainability targets are not met. Economic valuation using welfare values provides the mechanism to evaluate the projects or policy options that might offer solutions to correct for declining environmental performance. Its ability to capture all the value associated with environmental projects, including environmental existence values directly, is important to ensure we make decisions that are generating the maximum benefit to society. Figure A2.1 below provides advice on what values and economic tools are relevant for different types of common decisions with this in mind.

Importantly, BCA requires a clear articulation of the project counterfactual over a specified (usually future) time period to compare benefits and costs relative to what would (have) occurr(ed) in the absence of the project (or policy/management change). Without this understanding, one cannot properly inform the opportunity costs of investment, and hence cannot prioritise.

At the project-level, SEEA-EA is best if you want to:

- **Demonstrate in physical and monetary terms the impacts of a restoration project** as it occurs over time;

- **To compare detailed project-level accounts** (including the full range of co-benefits associated with restoration activities) with national accounts that do not have the same level of granularity;
- When undertaken recurrently, to **identify changes in the ecosystem** assets at the local level, and the flow of ecosystem services being provided

At the project-level, use economic valuation tools such as benefit-cost analysis if you want to:

- **Inform** an investment decision by establishing whether a project is worth undertaking (i.e. where benefits outweigh costs).
- **Prioritise / rank / compare** multiple projects or project designs to select the best projects for investment (i.e. to maximize net benefits).
- **Measure the contribution** that a project (or specific elements within that project, including the full range of co-benefits) makes to society (or to specific stakeholders in society).

Finally, it may not be important to estimate monetary-equivalent values to inform your decision in cases where you do not need to aggregate values for multiple ecosystem services into a single comparable metric. A key function of monetizing the contributions of ecosystem services is to ensure there is a systematic and comparable metric when aggregating and comparing over a range of services.

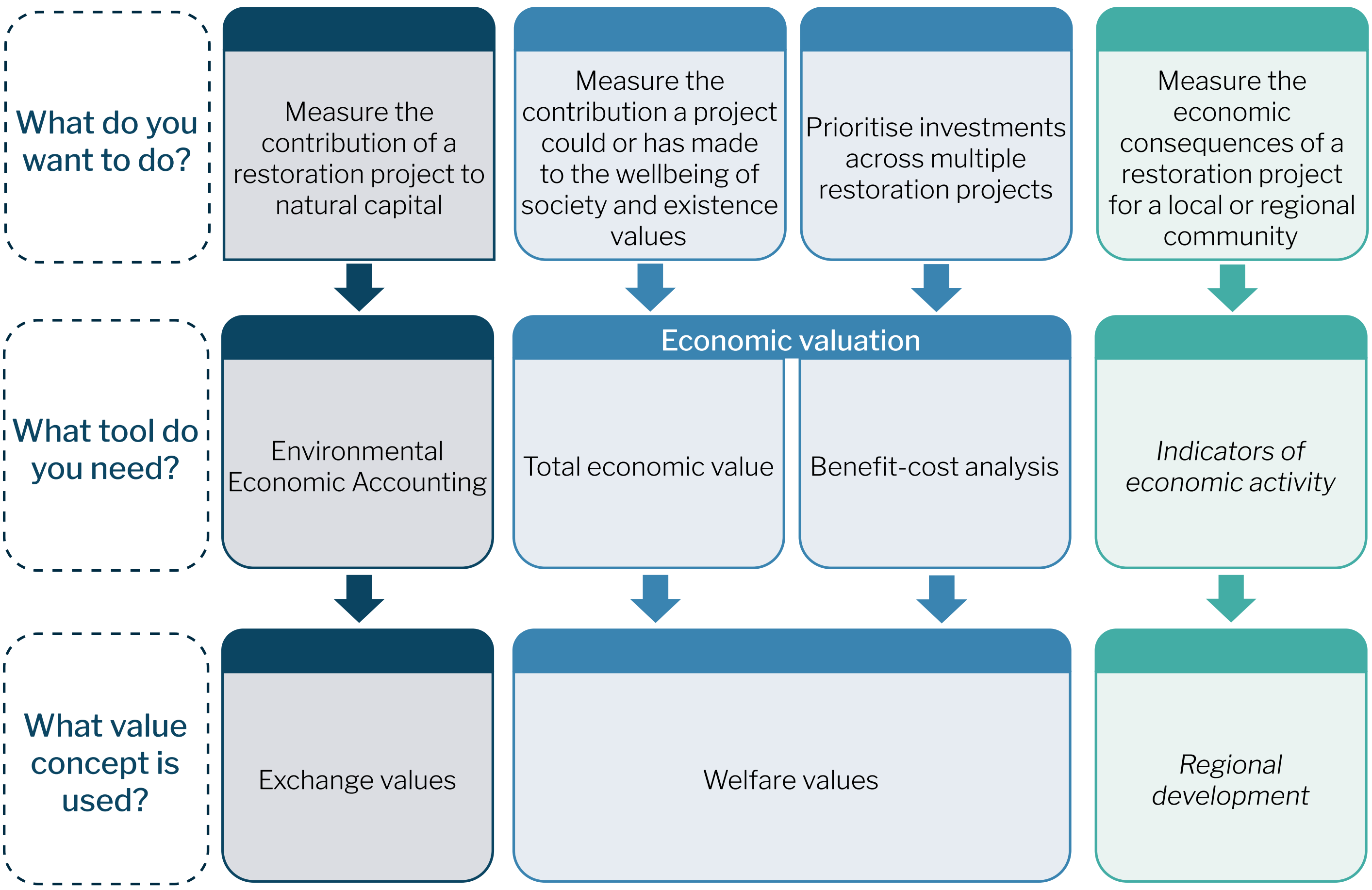

Figure A2.1: Common measurements required to support environmental decision making, and the appropriate valuation methods to provide each measurement.

Integrated economic assessment (benefit-cost analysis)

At the small-scale level (i.e., an environmental project) economic valuation rather than accounting tools are used to understand the contribution that a project could make (or has made, if retrospective) to the welfare of society. Benefit-cost analysis (BCA) estimates the benefits relative to the costs of undertaking a restoration project. This can be calculated for different projects or project design options to assess which project(s) deliver the largest net benefit.

BCA is based on the 'Total Economic Value' framework discussed above, where the welfare values (estimated as monetary benefits) associated with implementing the project are estimated relative to the welfare values associated with continuing with a 'business-as-usual' management approach. The costs of the project relative to business-as-usual are also factored in.

There are well established standard approaches to conducting BCA for projects[307]. An example is the Investment Framework for Economics of Water Sensitive Cities (INFFEWS) BCA Tool[308]. There are a number of steps for conducting benefit-cost analysis:

1. *Understand the issues and the context.*

   Projects can be complex and multifaceted when performing BCA. Being clear about what the potential projects are and clearly defining each project in a high level of detail is crucial. The specification of benefits can be highly flexible, with various types of benefit likely to emerge.

2. *Define project alternatives and baseline.*

   It is a crucial step to explicitly define the without-project scenario (considered as baseline), and identify credible and viable alternative policy options to be analysed (i.e., the with-project scenario(s)). Defining the without-project scenario and using it as the baseline for measuring the relative benefits from projects is vital to transparently analyse the impact of the project(s) under consideration.

3. *Identify types of benefits and costs, who benefits and who bears the costs or is adversely affected.*

   Identifying the various types of costs and benefits to all people affected by the project requires careful thought and guidance from a client. It is important to identify the costs and benefits for with- and without-project scenarios. The without-project should be clearly defined and forward-looking, recognizing that the current situation does not necessarily reflect what the future situation would be due to existing trends that may alter baseline conditions over time, even if new projects are not implemented.

4. *Quantify the benefits and costs, monetize them as far as possible. Estimate project risks and people's adoption of changes promoted by the project.*

   When quantifying costs and benefits, BCA should include all relevant information that can affect a decision. Where relevant, monetize all the relevant benefits and cost data to support the quantitative analysis. For example, it should include non-use values using appropriate methodologies that can estimate them as willingness to pay. Another essential aspect of the process is proper identification and quantification of project risks, including the risks of unfavourable technical, social, political, financial or management outcomes. If a project's success also relies on buy-in or actions to be taken by certain stakeholders, then the likelihood of adoption also needs to be considered.

5. *Discount future costs and benefits to obtain present values.*

   As most of the costs and benefits of a given project occur over time, and their value depends on when they are received,

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307 Boardman A. E. et al. (2018). Cost-benefit analysis: concepts and practice (Fifth). Cambridge University Press.

308 Pannell, D.J. (2019). INFFEWS Benefit: Cost Analysis Tool: Guidelines. Cooperative Research Centre for Water Sensitive Cities, Melbourne.

discounting is crucial to BCA. The need to discount future monetary values reflects that there is the opportunity cost of choosing to invest in one project, and not investing in something else. All monetised values incurred in the future need to be discounted back to present values. The analyst may use monetary value at the future date (the 'nominal' value) which accounts for inflation over time, or in terms of their current dollar value (the 'real' value) at time zero which avoids the need to account for future inflation.

6. *Compute decision metrics: Net Present Value and Benefit: Cost Ratio.*

The two common standard decision criteria for BCA are the net present value and benefit: cost ratio. The net present value of a project equals the present value of benefits minus the present value of costs with a positive value suggesting that the project is efficient (i.e., that it improves aggregate welfare or wellbeing). The benefit: cost ratio divides the present value of benefits by the present value of costs. In calculating the benefit: cost ratio, it is important to only place the project-constrained costs in the cost denominator, including things like the capital and operational expenses required to deliver the project. Other costs that are not relevant to the project budget, such as negative impacts that a project may cause for the environment or stakeholders who are external to the administering organisation, can be included in the numerator (i.e., these are effectively represented as 'negative benefits').

The correct decision metric should be selected to reach a recommendation. If the BCA is an evaluation of one project, and just needs to establish if it is worthwhile doing, either net present value or the benefit: cost ratio can be used.

If the BCA is comparing and prioritizing multiple projects, different metrics should be used for different circumstances. For most decisions there will be a budget constraint that dictates available resources for projects. If projects are mutually exclusive (doing one project rules out doing another), net present value should be used to rank projects. If projects are independent (multiple projects could be implemented), the benefit: cost ratio should be used. In the unlikely event that there is an unlimited budget, either metric can be used.

7. *Address uncertainty, including sensitivity analysis.*

Given there may be considerable uncertainty about estimates of costs and benefits, sensitivity analysis provides information about how changes in different variables will affect the overall costs and benefits of the project. This approach tests whether the uncertainty over the value of certain variables matters and identifies critical assumptions.

8. *Report results.*

The final step is to formally report the BCA results. The report should generate both contextual and quantitative information about the project and include results of net present values and benefit: cost ratios, in addition to sensitivity analysis.

Some useful links to additional BCA resources are provided below.

- A Guide for the Assessment of the Costs and Benefits of Sustainability Certification. https://www.unep.org/resources/report/guide-assessment-costs-and-benefits-sustainability-certification
- Australian Government, Department of the Prime Minister and Cabinet, Office of Best Practice Regulation. https://oia.pmc.gov.au/sites/default/files/2021-09/cost-benefit-analysis.pdf
- Commonwealth of Australia (2006). Handbook of Cost-Benefit Analysis, Financial Management Reference Material No. 6, Department of Finance and Administration, Canberra: https://www.atap.gov.au/sites/default/files/Handbook_of_CB_analysis.pdf
- Government of United Kingdom, The Green Book (2020). https://www.gov.uk/government/publications/the-green-book-appraisal-and-evaluation-in-central-governent/the-green-book-2020
- NSW Treasury (2023) Guidelines: Cost-Benefit Analysis. https://www.treasury.nsw.gov.au/finance-resource/guidelines-cost-benefit-analysis

Glossary

Glossary of relevant Ecosystem Services from SEEA*

*adapted from SEEA Table 6.3

Ecosystem Service as described in case studies	Ecosystem Service		Description
Provisioning services			
Fisheries biomass provisioning service	Biomass provisioning services	Wild fish and other natural aquatic biomass provisioning services	Wild fish and other natural aquatic biomass provisioning services are the ecosystem contributions to the growth of fish and other aquatic biomass that are captured in uncultivated production contexts by economic units for various uses, primarily food production. This is a final ecosystem service.
Fisheries nursery services	Other provisioning services		Nursery population and habitat maintenance services are the ecosystem contributions necessary for sustaining populations of species that economic units ultimately use or enjoy either through the maintenance of habitats (e.g. for nurseries or migration) or the protection of natural gene pools. This service is an intermediate service and may input to a number of different final ecosystem services including biomass provision and recreation-related services.
Regulating and maintenance services			
Flood control services	Flood control services	River flood mitigation services	River flood mitigation services are the ecosystem contributions of riparian vegetation which provides structure and a physical barrier to high water levels and thus mitigates the impacts of floods on local communities. River flood mitigation services will be supplied together with peak flow mitigation services in providing the benefit of flood protection. This is a final ecosystem service.
Carbon sequestration & emissions	Global climate regulation services		Global climate regulation services are the ecosystem contributions to reducing concentrations of GHG in the atmosphere through the removal (sequestration) of carbon from the atmosphere and the retention (storage) of carbon in ecosystems. These services support the regulation of the chemical composition of the atmosphere and oceans. This is a final ecosystem service.
Water quality	Water purification services (water quality regulation)	Retention and breakdown of nutrients Retention and breakdown of other pollutants	Water purification services are the ecosystem contributions to the restoration and maintenance of the chemical condition of surface water and groundwater bodies through the breakdown or removal of nutrients and other pollutants by ecosystem components that mitigate the harmful effects of the pollutants on human use or health. This may be recorded as a final or intermediate ecosystem service.
Cultural services			
Cultural services – recreation and existence values	Recreation related services		Recreation-related services are the ecosystem contributions, in particular through the biophysical characteristics and qualities of ecosystems, that enable people to use and enjoy the environment through direct, in-situ, physical and experiential interactions with the environment. This includes services to both locals and non-locals (i.e., visitors, including tourists). Recreation-related services may also be supplied to those undertaking recreational fishing and hunting. This is a final ecosystem service.
Cultural services – First Nations Values	Spiritual, artistic and symbolic services		Spiritual artistic and symbolic services are the ecosystem contributions, in particular through the biophysical characteristics and qualities of ecosystems, that are recognised by people for their cultural, historical, aesthetic, sacred or religious significance. These services may underpin people's cultural identity and may inspire people to express themselves through various artistic media. This is a final ecosystem service.

Glossary of terms

Term	Definition
ABARES	Australian Bureau of Agricultural and Resource Economics and Sciences.
Abiotic	Not from living organisms, only in the physical or chemical realm.
Acid Sulphate Soils (ASS)	Naturally occurring soils and sediments containing iron sulphides, most commonly pyrite. When ASS are exposed to air the iron sulphides in the soil react with oxygen and water to produce a variety of iron compounds and sulfuric acid. Initially a chemical reaction, the process is accelerated by soil bacteria.
Activities	Activities that occur in or near ecosystems that have impacts on the system, generally with economic benefits (for example, fishing).
Above Ground Biomass (AGB)	Living vegetation above the soil, including stem, stump, branches, bark, seeds, and foliage.
Annual Average Damage (AAD)	Calculated equivalent annual equivalent expense if hazard damages occurred evenly through time.
ArcMap	Main component of Esri's ArcGIS suite of geospatial processing programs, and is used primarily to view, edit, create, and analyse geospatial data.
Annual Exceedance Probability (AEP)	The probability (measured as a percentage) that a given rainfall total accumulated over a given duration will be exceeded in any one year.
Australian height data (AHD)	The Australian Height Datum (AHD) is the official national vertical datum for Australia and refers to Australian Height Datum 1971 (AHD71; Australian mainland) and Australian Height Datum (Tasmania) 1983 (AHD-TAS83).
Assets	A store of value from which economic benefits and ecosystem services are expected to flow over time. Assets encompass a broad range of entities, including: • Economic Assets (SNA): Tangible or intangible entities, owned or controlled by institutional units, from which economic benefits can be derived through their use, rental, or sale. • Ecosystem Assets (SEEA-EA): Specific spatial areas containing a combination of natural and cultivated components that function together as an ecological unit. Ecosystem assets are not necessarily owned by institutional units but are recognized for their capacity to generate a flow of ecosystem services, contributing to societal well-being and economic activity
Biodiversity	The diversity of life found within an area.
Biomass	The mass of biological matter, generally expressed in kg or t.
Biotic	Produced from living organisms.
Blue Accounting Model (BlueCAM)	A model used to estimate carbon stocks in a wetland ecosystem.
Blue Carbon Ecosystems	Ecosystems that contain blue carbon, which is stored atmospheric or oceanic carbon.
Carbon sequestration	The process of capturing and storing atmospheric carbon dioxide, often mitigating greenhouse emissions.
Carbon Estimation Area (CEA)	A stratum of the Project Area; land which is homogenous for the purpose of abatement calculations, has consistent biophysical characteristics and is established and managed in a consistent way. CEAs may be defined by a single CEA Polygon or, where a specific method allows, more than one CEA Polygon (see Split CEA).
Coastal protection	Physical protection provided by habitats to human developments.
Compositional state	A category of the SEEA-EA Ecosystem Condition Typology. The composition of an ecosystem, usually referring to plant or animal communities and their diversity. Biodiversity related measures would be recorded in this category
Conceptual model	Simplified flow chart outlining interactions between different factors relevant to the system examined.
Country	Country is the term used to describe the lands, waterways and seas to which First Nations' People are connected. The term contains complex ideas about law, place, custom, language, spiritual belief, cultural practice, material sustenance, family and identity.
DCCEEW	Commonwealth Department of Climate Change, Energy, the Environment and Water.

Term	Definition
Digital Earth Australia (DEA) sandbox	The Digital Earth Australia (DEA) Sandbox is a learning and analysis environment for getting started with DEA data and our Open Data Cube.
Digital elevation model (DEM)	A Digital Elevation Model (DEM) is a representation of the bare ground (bare earth) topographic surface of the Earth excluding trees, buildings, and any other surface objects.
Ecosystem condition	The quality of the ecosystem measured in terms of its abiotic, biotic and landscape/ seascape characteristics. Successfully restored habitats should see their condition improve.
Ecosystem conversion	Amount of change in restored habitats before and after restoration activities
Ecosystem extent	Spatial area covered by an ecosystem, expressed in hectares (Ha), m^2 or km^2. Also 'size of ecosystem asset'.
Ecosystem service	The many and varied benefits to humans provided by the natural environment and from healthy ecosystems. For example, the fish they produce that are then consumed by fisheries.
eCognition	Trimble eCognition software is used by GIS professionals, remote sensing experts & data scientists to automate geospatial data analytics.
Ecotone	A transitional area of vegetation between two different plant communities, for example between saltmarshes and mangroves.
Environmental Economic Accounting (EEA)	A framework for organising statistical information to help decision-makers better understand how the economy and the environment interact.
Environmental economic account	Accounts used to value ecosystems, usually comprised of an ecosystem extent account and an ecosystem condition account.
Environmental economic accounting	Framework used to compile information linking environmental factors to economics.
Emissions trading register (ETR)	An online database that issues, records, and tracks the carbon units that are exchanged within market mechanisms or financed through Results-Based Climate Finance programs.
Emissions reduction fund (ERF) now ACCU Scheme	The ACCU Scheme offers landholders, communities and businesses the opportunity to run projects in Australia that avoid the release of greenhouse gas emissions or remove and sequester carbon from the atmosphere.
Fine benthic organic matter (FBOM)	Deposited on the stream bottom (i.e., fine benthic organic matter) can vary greatly between stream habitats (e.g. pools and riffles) and is a key food for deposit feeders (analogous to microphytobenthos).
First Nations ecosystem services	Services provided by natural habitats to First Nations people.
Fisheries biomass provisioning service	The animal product (e.g. fishes and crustaceans) produced from ecosystem services that is caught and sold by fisheries.
Flows	Broadly within the accounting context, flows include the movement of assets, resources, items, or values into or out of an entity or account over a specific period. They are responsible for the changes between opening and closing stocks. Ecosystem services are flows in the context of ecosystem accounting.
Food web	A more complicated version of a food chain that includes all feeding interactions between organisms in an ecosystem.
Functional state	The function of the community.
Geographic Information Systems (GIS)	Software systems used to process spatial information, to create maps, for example.
Global climate regulation	Activities, natural or human-caused, that help regulate the climate, generally through lowering atmospheric greenhouse gases.
Habitat maintenance	Services provided by natural habitats to themselves that are required for ecosystem function.

Term	Definition
Highest astronomical tide (HAT)	Defined as the highest level which can be predicted to occur under average meteorological conditions and any combination of astronomical conditions.
Hunter Valley Flood Mitigation Scheme (HVFMS)	State owned engineering work, designed to lessen the effects of flooding on both rural and urban areas, reducing flood damage by modifying flood behaviour.
Indigenous Protected Areas (IPA)	Areas of land and sea Country managed by First Nations groups in accordance with Traditional Owners' objectives.
Hydrodynamic regime	Patterns in water flow within or across an ecosystem, for example tidal patterns.
Indigenous cultural resource management (ICNRM)	ICNRM are activities undertaken by Indigenous individuals, groups and organisations across Australia constituted by caring for Country and which originate from the millennia old relationships between traditional Aboriginal and Torres Strait Islander societies and their Country. Cultural resource management such as hunting, gathering, burning, ceremony, knowledge sharing are actions taken to protect and maintain the health of Country and its people.
(Australian) Institute of Aboriginal and Torres Strait Islander Studies (IATSIS)	This is a national institute that provides advice on how to engage with Aboriginal and Torres Strait Islander communities, support recognition of ATSI culture, narrative and resurgence. They also provide formal advice on and protocols for how to conduct research with ATSI people.
Landsat	Earth observation satellite system run by NASA (digital remote sensed data).
Lowest astronomical tide (LAT)	Defined as the lowest level which can be predicted to occur under average meteorological conditions and any combination of astronomical conditions.
Light detection and ranging (LiDAR)	Also called 3D laser scanning, LiDAR is a method for determining ranges by targeting an object or surface with a laser and measuring the time for the reflected light to return to the received.
Modified Normalised Difference Wetness Index (MNDWI)	Uses green and short-wave infrared band pixel values to enhance open water features in GIS applications.
Modelling	Extrapolating patterns, either between known data points, or into the future.
Monetary accounts	Accounts that measure the value of ecosystems for society.
Monitoring	Repetitive assessments of habitat condition, usually conducted annually or every 5 years.
Mandingalbay Yidinji Aboriginal Corporation (MYAC)	Under the Native Title determination the Mandingalbay Yidinji Aboriginal Corporation was set up as the Registered Native Title Prescribed Body Corporate which holds the Native Title for National parks, other reserves and state forest areas near Cairns.
Natural State	In the Australian context this refers to the condition of an area before colonisation occurred.
Normalised Difference Vegetation Index (NDVI)	The data provides an overview of the status and dynamics of vegetation across Australia, providing a measure the amount of live green vegetation using grids and maps from satellite data. The satellite data comes from the Advanced Very High Resolution Radiometer (AVHRR) instruments on board the National Oceanic and Atmospheric Administration (NOAA) series of satellites that are operated by the US (http://noaasis.noaa.gov/NOAASIS/ml/avhrr.html).
Nearmap	Nearmap is an aerial technology company that provides frequently-updated high-resolution aerial imagery and location intelligence.
Non-market valuation (NMV)	Non-Market valuation is a set of techniques that aims at reflecting the economic value of changes, in the availability or quality, of goods and services that are not intended to be traded in the market.
Nursery population	Role of habitats for assisting the growth of young animals.
Object-based image analysis (OBIA)	A type of image analysis that groups cells into objects (i.e., vectors) based on their spectral, geometrical and spatial properties to partition and classify Earth observation data.

Term	Definition
Orthomosaic	The output from a process where a number of overlapping photos (e.g. from a drone or aerial camera) are stitched together with distortions removed to create a complete and continuous image representation or map of a portion of the earth.
Physical accounts	Accounts that measure the physical distribution of ecosystems, for example habitat extent or productivity.
Pools	Components of an ecosystem that can 'store' carbon.
Primary data collection	Information requiring boots on the ground at the site of restoration to assess, data not currently existing elsewhere.
Project scoping	Determining the size of the project, both in time and space.
QGIS	QGIS is a free and open-source cross-platform desktop geographic information system (GIS) application that supports viewing, editing, printing, and analysis of geospatial data.
Ramsar	The Ramsar Convention on Wetlands of International Importance Especially as Waterfowl Habitat is an international treaty for the conservation and sustainable use of Ramsar sites (wetlands). It is also known as the Convention on Wetlands. It is named after the city of Ramsar in Iran, where the convention was signed in 1971.
Reference level	The value of a variable at the reference condition, against which it is meaningful to compare past, present or future measured values of the variable.
Reference sites	Sites with similar habitat at the restoration site, used to assess relative changes as being caused by restoration activities.
Regional multipliers	An expenditure that leads to broader economic benefits, for example the value of 1kg of prawns caught leads to broader economic benefits from processing, transport etc.
Remote sensing	Process of detecting and monitoring the physical characteristics of an area using aircraft or satellites, without physically interacting with the habitat.
Remotely-piloted aircraft (RPA), or UAV	Aircraft flown without a person on-board, also called "drone" or Unmanned Aerial Vehicle (UAV).
Restoration project	A project aiming to undo damage caused by human activities within a given area, usually trying to revert to conditions pre-human influence.
SEEA	System of Environmental Economic Accounting A formal framework developed by the UN for valuing ecosystem services.
SEEA-EA	System of Environmental Economic Accounting Ecosystem Accounting.
SIMMR	A statistical package in R designed to solve mixing equations for stable isotopic data within a Bayesian framework.
Spatial coverage	Area covered by the project.
Spatial resolution	How easy is is to distinguish two neighbouring structures as separated, higher is usually better but comes at a cost of data maintenance issues. Usually expresssed in m or km.
Stable isotopes	Naturally-occuring elements (e.g. Carbon) that do not decay like radioisotopes.
Stakeholder	A stakeholder is either an individual, group or organisation that's impacted by the outcome of a project or a business venture. Stakeholders have an interest in the success of the project and can be within or outside the organisation that's sponsoring the project.
Statistical	Summarising numbers in a way that is objective.
Stocks	Natural resources or land, such as fish stocks.
Supply and use tables	Record flows of goods and services, including ecosystem services, between economic units and the environment, including ecosystems.

Term	Definition
Supratidal forest	Forest occurring on a tidal flat above the level of mean high water for spring tides, 'splash zone'. In Australia generally include *Melaleuca* and *Casuarina* trees.
Trophic enrichment factor (TEF)	A parameter reflecting the difference in isotopic ratio between a consumer's tissues and diet, used in isotopic ecology and paleoecology to track dietary habits.
Temporal coverage	Historical time across which data will be collected.
The Guide	A Guide to Measuring and Accounting for the Benefits of Blue Carbon Ecosystem Restoration, 2023.
Universal Transverse Mercator (UTM)	A map projection system for assigning coordinates to locations on the surface of the Earth.
Validation	Assessing the accuracy or uncertainty of higher-level remote sensing products with analytical reference data (such as corresponding ground and field measurements or using experts to verify).
Water purification service	Processes that increase the quality of the water, for example often reducing levels of pollutants.
Woody Vegetation Cover Fraction (WCF)	Vertical projection area of vegetation cover index used in remote sensing applications.
World Geodetic System (WGS)	A standard used in cartography, geodesy, and satellite navigation including GPS. The current version, WGS 84, defines an Earth-centered, Earth-fixed coordinate system and a geodetic datum, and also describes the associated Earth Gravitational Model (EGM) and World Magnetic Model (WMM).
Willingness to Pay (WTP)	The maximum price a customer or consumer is willing to pay for a product or service.

Project contacts:

Dr Paul Carnell

paul.carnell@deakin.edu.au

Kym Whiteoak

kym@canopyeco.com.au