Freshwater 2022

# Guidance for implementing the NPS-FM sediment requirements

Ministry for the Environment
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Te Kāwanatanga o Aotearoa
New Zealand Government

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**Feedback**

If you have feedback on this guidance, please email freshwater@mfe.govt.nz.

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# 1 Introduction

## 1.1 Erosion and sediment

Sediment is one of the most pervasive and significant contaminants in aquatic systems, contributing to degradation of ecosystem health and amenity values.

The erosion that gives rise to sediment takes various forms (table 1-1). In Aotearoa New Zealand, shallow landslides are the most common (Basher, 2013).

**Table 1-1: Types of erosion**

**Surficial erosion**: The loss of rain-dislodged soil particles through sheet-wash, rill and inter-rill processes.

**Shallow landslides**: Rain-initiated shallow landslides, typically about 1 metre deep and spanning 50–100 $m^2$.

**Earthflows**: The slow movement of soil and regolith (loose rock and dust above a layer of bedrock) along marginal shear planes (eg, erosion terrains underlain by crushed mudstone and argillite lithology).

**Gully erosion**: Erosion initiated by water flowing into gully heads and scouring out the drainage channel.

**Bank erosion**: Stream bank erosion induced by gravity-driven collapse, current scour, or a combination.

Aotearoa New Zealand’s mountainous, tectonically active landscapes, combined with plentiful rainfall, make erosion and the loss of sediment to streams a natural process. However, human activities have accelerated this.

### Adverse effects

Excess fine sediment[1] (ie, more than would usually result from natural processes in an unmodified catchment) can have adverse effects on water quality and freshwater ecosystems, by making water turbid and stream beds muddy.

Most forms of freshwater life evolved to cope with the natural sedimentation rates but, in the past 700 years, sedimentation has increased as natural forest cover has been reduced from 80 to 90 per cent of the land area to around 30 per cent.

When water bodies receive more sediment than they can remove or disperse, their natural character may be altered, with negative effects on their plant, invertebrate and fish populations.

High concentrations of suspended fine sediment (turbidity) can reduce water clarity, limit light penetration and release nutrients (eg, phosphorus). This can change the distribution and abundance of aquatic plants and algae, and some fish species.

[1] Fine sediment has a grain size less than 2mm in diameter.

Deposited fine sediment can change benthic (stream bottom) communities, eliminating some invertebrates, algae and micro-organisms while favouring others. This can alter the food chain and reduce the diversity of fish species feeding there, particularly in the stream's lower reaches.

In the presence of elevated nitrogen levels (eg, from farmland), the release of phosphorus from suspended and deposited sediment can trigger the growth of algal blooms.

Excessive sediment can also reduce the water's suitability for human uses, such as drinking and swimming.

### Managing fine sediment

The broad goal of management is to limit fine sediment discharges or loads, as much as possible, to something approximating natural levels, and to remove and prevent excess accumulations of deposited sediment.

While we cannot control the forces of nature, it is possible to control the human activities that make soil more vulnerable to natural disturbance. These include livestock grazing on slopes and stream banks, vegetation clearance, ploughing and earthworks. These activities are associated with a range of industries, including farming, forestry, urban development, construction, transport and mining.

Among the high-risk activities that can trigger or exacerbate fine sediment loss are stock accessing waterways, harvesting trees, intensive winter grazing, hill country forage cropping, building tracks, roads and culverts, cultivating soil, and poor drainage management.

## 1.2 Council responsibilities

The main responsibility for **erosion and sediment control (ESC)** lies with regional and unitary councils (hereinafter councils). They are charged with managing freshwater under the Resource Management Act 1991 (RMA) and have specific ESC responsibilities under the National Policy Statement for Freshwater Management 2020 (NPS-FM).

### Attributes

The NPS-FM requires councils to manage the impacts of sediment on rivers and their downstream receiving environments by controlling:

- *suspended fine sediment* (in all rivers, measured as visual clarity, or as turbidity converted to visual clarity) and
- *deposited fine sediment* (in wadeable and naturally hard-bottomed rivers, measured as the percentage of the stream-bed area covered with deposited fine sediment).

These two sediment variables are referred to as attributes in the NPS-FM. Attributes are measurable characteristics of freshwater that can be described numerically or by narrative, or both.

To manage these, councils must decide how much sediment is acceptable, and put plans in place to reduce it and keep it within that level. See National Objectives Framework attributes guidance.

## Target attribute state (TAS)

The first step for councils is to set a target attribute state (TAS) for each attribute, in consultation with their communities and iwi. The TAS must be set at a level that either improves each attribute or maintains it at, or better than, its **baseline state**,[2] or – if the baseline is of lower quality than the **national bottom-line (NBL) –** at a state that is no worse, and ideally better, than the NBL (see appendix A). See National Objectives Framework TAS guidance.

## Limits on resource use

Councils must then set rules in their regional plans placing limits on resource use to ensure that the **visual clarity TAS** is met. These limits may apply to any activities that, in the council's assessment, pose a sedimentation risk to freshwater and may be expressed as a **land-use control** (such as a control on the extent of an activity), an **input control** (such as livestock numbers), or an **output control** (such as a volume or rate of discharge). See National Objectives Framework limits guidance.

## Action plans

Councils are also required to develop action plans, where appropriate, to achieve their deposited fine sediment TAS. This involves restoring selected soft-bottomed[3] streams to a hard-bottomed state, if that was their former natural state, and maintaining that state. Setting limits and developing action plans requires an understanding of catchment characteristics and, in particular, sediment loads and their sources. However, imperfect information is not an acceptable reason to delay limit-setting and action plans. See National Objectives Framework action plan guidance.

## Monitoring

To track progress, councils must set up a monitoring system for the two sediment attributes. In addition, though not a statutory requirement, monitoring sediment sources and relevant land use activities will contribute to more effective resource use limits and action plans. See National Objectives Framework monitoring guidance.

# 1.3 Important concepts and requirements

The NPS-FM sets a new direction in freshwater management that requires a fundamental rethink of how we treat the freshwater environment and how we prioritise its values and uses. Under this new approach, the resource use limits and action plans for controlling freshwater sedimentation must give effect to the following important core concepts and requirements:

- the Te Mana o te Wai hierarchy of obligations (TMOTW)
- national bottom-lines (NBLs)
- engagement with communities and active involvement of tangata whenua
- "ambitious and reasonable" timeframes.

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[2] Baseline state is defined in clause 1.4 of the NPS-FM. For more detail, see section 2 of this document.

[3] Soft-bottomed means a site where the bed has a greater than 50% coverage of deposited fine sediment.

These are summarised here and explained more fully in other NPS-FM guidance documents, eg, the National Objectives Framework guidance and Te Mana o Te Wai factsheet (Ministry for the Environment, 2022; Ministry for the Environment and MPI, 2022).

### 1.3.1 The Te Mana o te Wai Hierarchy – water comes first

The Te Mana o te Wai (TMotW) hierarchy of obligations underpins the new freshwater management system that all councils must implement by 2024. Its **top priority is the health and wellbeing of water bodies and freshwater ecosystems.** This must be assured before enabling human use of a water body. The second priority – using water for human health needs (eg, drinking) – may only occur where this will not jeopardise the first priority. The third priority – using water to meet human economic, cultural, social, and recreational needs – can only be provided for if both the ecological and human health priorities are not jeopardised.

TMotW means that it is **no longer appropriate to 'balance' water priorities by making trade-offs** which favour human use of water at the expense of reduced water quality or quantity. Best practice for giving effect to TMotW is to take a precautionary, 'water comes first', approach when setting targets, limits on resource use, and action plans. This will best meet the NPSFM requirement for no loss of water quality (and improvement where necessary).

### 1.3.2 National bottom lines (NBLs) – minimum thresholds

The NPS-FM sets clear national bottom lines (NBLs) for various waterbody attributes. Those for sediment are set out in tables 8 (suspended fine sediment) and 16 (deposited fine sediment) of appendix 2A and 2B of the NPS-FM. (See appendix A here.)

There is a risk of NBLs being perceived as recommended targets. They are not. They are **minimum thresholds that water quality must be raised to and must not decline to.** Councils must improve water attributes to at least their NBL and must maintain or improve those that are in a better state than their NBL. The most critical point is that it is not acceptable to allow water bodies to decline from their current state, even those that are well above the NBL.

### 1.3.3 Fair participation – for communities and tangata whenua

Councils must engage widely and transparently so that **everyone can participate fairly when setting freshwater outcomes, objectives and targets**. This includes tangata whenua, catchment communities, and anyone in the wider community with an interest in the region's freshwater management. Six principles for tangata whenua involvement in freshwater management are set out in clause 1.3(4) of the Te Mana o te Wai section of the NPS-FM.

### 1.3.4 Timeframes – sooner rather than later

It will often be impractical to achieve a TAS immediately, so timeframes in long-term visions must be "both **ambitious and reasonable** (for example, 30 years after the commencement date)" – that is, difficult but not impossible to achieve.

The NPS-FM recognises that achieving freshwater goals can take time as communities and resource users adjust and make changes. If this requires business transitions, the timeframe should allow for **orderly transitions within a clear and timely deadline**. To maintain steady progress, intermediate targets can be set, with achievable interim timeframes. Some transitions, such as land use change, may require longer timeframes, but these should not be unduly prolonged. Councils should signal any need for land use changes early, to enable efficient transition planning and avoid land users making unsustainable investment decisions.

# 2 Overview of this document

To assist councils, the Ministry for the Environment (the Ministry) commissioned the National Institute for Water and Atmospheric Research (NIWA) to draft guidance on:

- Linking the current sediment attribute states and the target sediment attribute states to catchment sediment loads.
- Setting limits on resource uses to achieve the TAS for suspended fine sediment (and therefore visual water clarity).
- Developing action plans to achieve the TAS for deposited fine sediment.
- Managing catchments to mitigate fine sediment effects in estuaries (to address the NPS-FM requirement to have regard to the effects of sediment in downstream receiving environments).

The guidance is underpinned by the premise that **instream visual clarity (and turbidity**, a commonly measured proxy or surrogate for visual clarity) and **deposited fine sediment cover** are influenced by **catchment mean annual fine sediment delivery** and that this, in turn, is influenced by **resource uses** (eg, soil disturbance, erosion-inducing land uses) that councils can identify and manage appropriately.

Before preparing the guidance, NIWA organised a workshop with representatives from regional councils, key industry sectors, and the Ministry. Workshop participants discussed monitoring sediment attributes, setting limits on resource use, and developing action plans to reach TASs.

The resulting NIWA guidance, as edited by the Ministry, is in sections 3–9 of this document. The Ministry prepared section 1 and made editorial contributions to all other sections. The Ministry and NIWA prepared section 2 and NIWA prepared sections 3–9. Manaaki Whenua Landcare Research (MWLR) contributed to sections 1–9.

## 2.1 Scope

This guidance addresses the four topics that the Ministry requested of NIWA. It is intended to be a high-level pointer to methods and information that will be of use in developing ESC measures, plan rules, resource use limits and action plans.

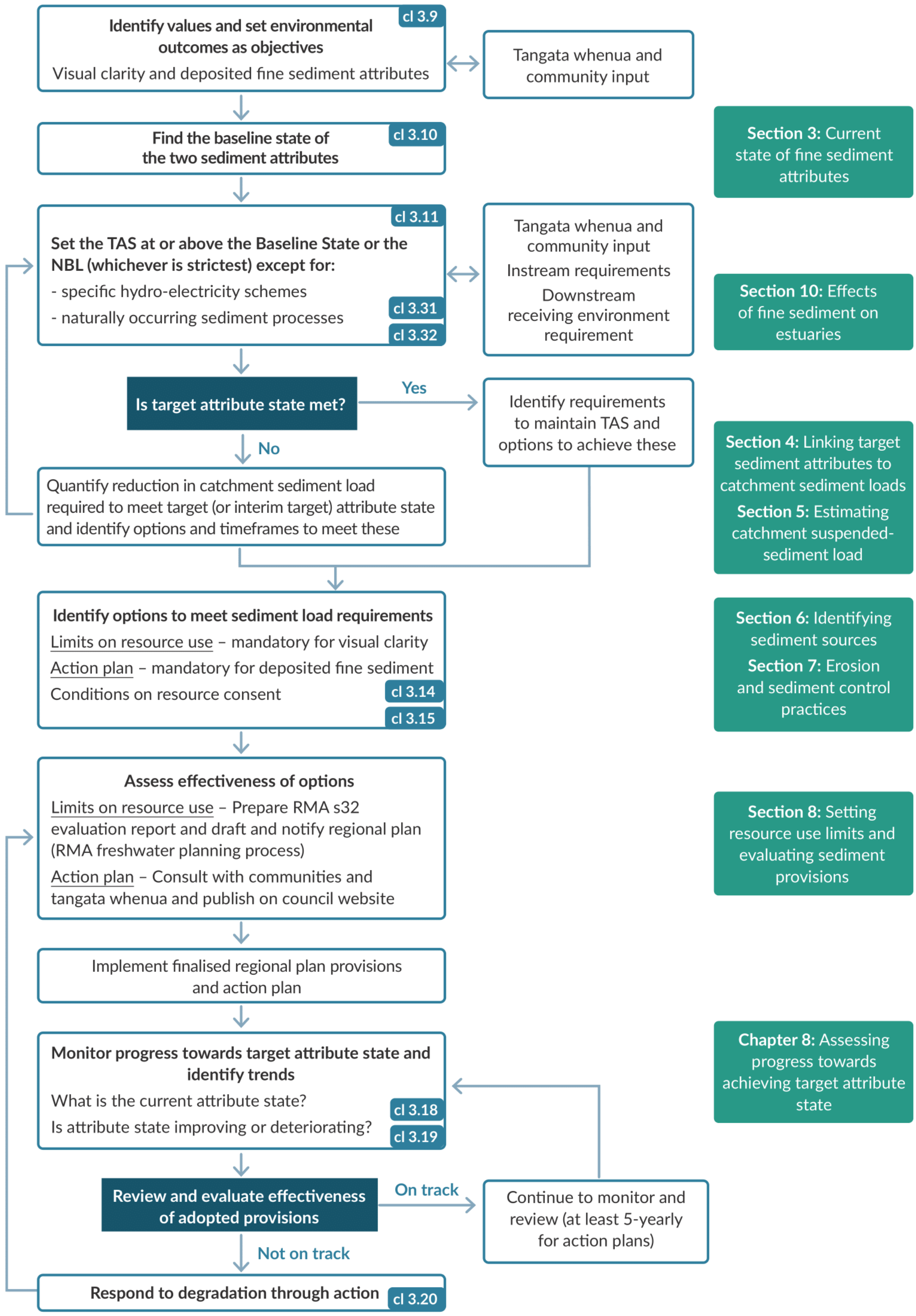
It does not replace the need to tailor these measures, rules, limits and action plans to local conditions, based, wherever possible, on high-quality regional and catchment level data, and on full engagement with the public, the stakeholders and tangata whenua – while also noting, however, that the absence of high-quality data is not a justification for delaying or deferring planning measures and actions.

The guidance does not address the detailed planning requirements for sediment management (eg, how to set resource use limits and how to carry out RMA section 32 cost–benefit analyses in line with the TMotW hierarchy). For these topics, see the National Objectives Framework guidance (Ministry for the Environment, 2022) and the Te Mana o te Wai factsheet (Ministry for the Environment and MPI, 2022).

## 2.2 The workflow – key steps and questions

The starting point for this guidance (top left of figure 2-1) is that environmental outcomes, expressed as objectives for a regional plan, have been established for ecosystem health (and any other relevant freshwater values). These outcomes are at the scale of a freshwater management unit (FMU) for both of the fine sediment attributes.

**Figure 2-1: Decision tree for implementing the NPS-FM fine sediment requirements**




The blue boxes refer to clauses in the NPS-FM 2020. The green boxes outline the sections in this guidance. For clarity, not all potential feedback loops are depicted. The decision-making shown here does not occur in isolation from other freshwater values, attributes, and national policy requirements. Tangata whenua are to be involved in all freshwater management and decision making.

The workflow in figure 2-1 shows key steps that councils should follow to give effect to the NPS-FM requirements for managing fine sediment. Most of these stages can be represented by the following simple questions.

## 2.2.1 What is the baseline state of each attribute?

The **baseline state** is defined in the NPS-FM (clause 1.4) as the attribute's *best* state out of the following:

(a) when first identified by a regional council

(b) when the council set an objective for it under the NPS-FM 2014 (as amended in 2017)

(c) on 7 September 2017.

This differs from the **current state***,* which is its state at any specified reference point, whether initially (when setting the baseline), now, or in the future (when reviewing progress toward the TAS). The states are assessed by analysing the results of river monitoring or, where insufficient data exist, by modelling.

## 2.2.2 What is the chosen target attribute state (TAS)?

Councils must determine the TAS with tangata whenua and community input. With few exceptions,[4] the TAS must be at or above the NBL (if the current state is worse than the NBL) or at, or above the baseline state (if the current state is better than the NBL).

Consider both instream and downstream ecosystem health requirements when setting the TAS. Downstream environments, such as estuaries, may be more sensitive to sediment than the rivers that flow into them.

## 2.2.3 Does the baseline state meet the TAS?

If the answer is 'yes', the council must set rules to ensure that the baseline state is maintained at the TAS level from a specified date (3.11(5)(a)). That is, no future decline or overallocation is possible and cumulative environmental effects are avoided.

If the answer is 'no', the council must identify how much of a reduction is needed for the catchment sediment load to meet the TAS. If it is a significant reduction, the council will need to evaluate how achievable it is over a reasonable but ambitious timeframe (eg, 30 years), setting different resource use limits and, if necessary, interim target states (of up to 10 years).

If it appears that the TAS will not be achievable within that timeframe, the council may need to consider adjusting the TAS. In doing so, it must ensure that the revised TAS still improves the attribute's current state, and still equals or betters the baseline state or the NBL – whichever is higher quality.[4]

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4 There are two exceptions where the TAS can be set below the NBL. These relate to specific hydro-electricity generation schemes (clause 3.31 of the NPS-FM) and to all or part of a water body in which the attribute is affected by naturally occurring processes, such as suspended glacial flour (clause 3.32). However, in both instances, the TAS must still improve on the current attribute state as far as is practicable.

### 2.2.4 What ESC options are available?

To set appropriate resource use limits, the council needs to know what ESC options are available. To do this, determine the catchment fine sediment load, and identify and understand fine sediment sources.

ESC options will differ according to the sources, their erosion processes, their spatial distribution, and their contribution to the load. Viable ESC options will target sources that are human influenced and make a significant contribution to the load.

### 2.2.5 Which ESC options are best for setting resource use limits and action plans?

After identifying a range of ESC options, the next step is to assess how effective and efficient they will be when implemented through resource use limits and action plans.

Resource use limits may take varying forms (eg, land use, input, or output controls, or a combination of these) but, in the case of fine suspended sediment, they are mandatory under clause 3.14 of the NPS-FM. They can also be used to manage deposited fine sediment cover but, as a minimum, deposited sediment must be managed through an action plan. An action plan may contain regulatory and non-regulatory measures.

Conditions on resource consents (eg, for discharges or activities that disturb the banks or beds of streams) are another mechanism councils can use to meet and maintain fine sediment TASs.

For all proposed measures and plan provisions, section 32 of the RMA requires an evaluation report, to assess the most effective and efficient options as well as their socio-economic costs and benefits. The section 32 report will be notified with the regional plan as part of the RMA freshwater planning process, which includes public submissions, a hearing and appeals.

For more guidance on s32 evaluation, see Ministry for the Environment (2022).

### 2.2.6 Which plan rules will ensure that anthropogenic sediment loads are controlled?

Resource use limits must be specified in regional plan rules, as set out in clauses 3.12(1)(a) and 3.14 of the NPS-FM. The aim of these rules is to restrict unnatural,[5] or anthropogenic, sediment loss to levels that will allow the TASs to be achieved.

For best practice, the rules should not devolve limit-setting or other key requirements to the consenting or farm-planning processes. The resource use limits should be clearly set in the regional plan. The rules should ensure that the limits for each resource user do not collectively exceed the catchment's sustainable sediment load (ie, the load that will allow TASs to be met).

[5] Sediment loss is unnatural when it exceeds what would usually result from natural processes in an unmodified (typically, forested) catchment. Usually, this is due to human influence on vegetation cover and land use. The aim of these plan rules and limits is to reduce this anthropogenic sediment loss.

To be effective, limits should be applied to three critical parameters of resource or land use:

- **farming practices** – many on-farm practices, such as land use choices and stock management decisions, can influence sediment loss, but farm practice *limits*, such as requiring good management practices (GMPs), are not the same as resource use limits, so would be insufficient on their own to achieve the TASs unless they were directly linked to specific resource uses that cause sediment loss (and can be monitored and enforced).
- **land use intensity** – this refers to the *intensity of sediment-generating activity*, such as the impact of high stocking rates, and may be managed by rules limiting the stocking rate on different types of erosion-prone land and in high-risk catchments.
- **land-use extent** – this refers to the *area of land affected by sediment-generating activities*, such as grazing and treading, and may be managed through rules that restrict certain types of land use on erosion-prone soils and in high-risk catchments.

  For example, intensive winter grazing (IWG) on high-risk soils will, even with GMP standards being applied, discharge a residual (sometimes a large residual) amount of sediment in high-rainfall events. To set an effective limit, councils would need to restrict the extent of intensive winter grazing in a catchment.

**To set resource use limits** councils will need to take these steps:

1. Identify current sediment loads and, using the best available information, differentiate the component that is natural from the component that is anthropogenic.
2. Identify and rank the different sources of anthropogenic sediment according to their load.
3. Identify the anthropogenic sediment load that will meet the desired TAS.
4. Identify the complete set of limits and actions that will 'hold' the current sediment loads.
5. Where reductions in the sediment load are required to meet TASs, set out the complete set of actions (on land-use practices, extent and intensity) needed to meet the reductions.
6. Establish a rules framework and cascade to:
   - prohibit use above and beyond the limit
   - set a transition pathway to address overallocation, that provides for progressive sediment reductions from the current state.

**The plan rules** may include 'plan B' criteria for implementing further resource use limits and other actions if adequate progress is not being made towards the TAS or to avoid overallocation (as defined in clause 1.4 of the NPS-FM). These could take the form of conditional rules that would be invoked if certain water quality thresholds were crossed, or specific milestones were missed.

Having a 'plan B' setting in the rules would allow for rapid adaptive management, without the need for a plan change, in situations where freshwater quality is declining or is stagnating below the NBL.

Another thing to consider when developing plan rules is how the they will interact with other regulatory requirements affecting land use, such as other regional rules, district plans, and other national direction.

For more guidance on setting rules and limits refer to the National Objectives Framework guidance (Ministry for the Environment, 2022).

### 2.2.7 How best to monitor progress towards the TAS?

Once the council has finalised and implemented the regional plan and action plan provisions, the next step is to monitor progress towards the target attribute state. This is vital for timely and effective management.

Under clause 3.20 of the NPS-FM, actions are to be proportionate to the likelihood and magnitude of the trend, the risk of adverse effects on the environment, and the risk of failing to reach the TAS. For this, regular monitoring is vital.

Clause 3.29(c) in the NPS-FM requires councils to provide baseline information to track over time the cumulative effects of activities. The NPS-FM also requires instream monitoring, to assess attribute states (appendix A) and long-term trends.

After an appropriate lag time for the controls to take full effect, the monitoring results will signal whether or not sediment levels are moving away from the baseline state and, if so, whether they are moving in the right direction.

The council must investigate any deteriorating trends in visual clarity or deposited fine sediment, interpret what is happening, and why, and respond effectively. The monitoring that is necessary for this will occur both in the water and on the land.

Monitoring the waterbody provides the necessary information on clarity and deposition but, without information on sediment sources, councils cannot interpret and respond effectively. Land-use monitoring is therefore needed to identify sediment sources and track their changes over time (eg, in the extent of high-risk activities like IWG).

Unless the source is a natural process, the council must take action to halt or reverse any degradation. This may involve enforcement action against noncompliant resource users, initiating on-the-ground ESC interventions (eg, tree planting), activating any regional plan adaptive management provisions, changing the regional plan, amending an action plan, or preparing a new one.

## 2.3 Section themes

The following sections of this guidance address the workflow steps shown in figure 2-1 above.

Section 3: **Monitoring the current attribute state** for suspended and deposited fine sediment, focusing mainly on the use of continuous turbidity monitoring as a surrogate for visual clarity.

Section 4: **Linking sediment attributes to sediment load**, so that the TAS can be converted to absolute or relative (ie, proportional) sediment load reductions for setting resource use limits.

Section 5: The principal approaches for **quantifying a catchment's suspended sediment load**.

Section 6: Tools and approaches for **locating and quantifying major sediment sources**.

Section 7: **Documented ESC practices**, including efficiency and timeframe to effectiveness.

Section 8: **Monitoring progress**, by quantifying the rate and direction of change in TAS.

Section 9: **Management of sediment in estuaries,** including the measurement and modelling of base state and the impacts of sediment accumulation processes.

# 3 What is the current state of fine sediment attributes?

This section addresses fine sediment attribute monitoring for suspended fine sediment and deposited fine sediment cover to assess NPS-FM compliance.

It includes direct measurements of visual clarity, continuous turbidity monitoring as a surrogate for visual clarity, and measuring the extent to which fine sediment covers the visible streambed.

See NPS-FM: appendix 2A and 2B, tables 8 and 16.

Managing fine sediment in rivers, under the NPS-FM, requires knowledge of the current state of two sediment-related attributes: suspended and deposited fine sediment.

NPS-FM table 8 (appendix 2A) and table 16 (appendix 2B), reproduced here in appendix A, set out four numerical state bands ('grades') for these two attributes (A to D). They also establish the C/D band boundary as the national bottom line (NBL) or minimum acceptable state.

These attribute bands have been mapped across the national digital stream network (Zammit, 2017) according to classifications for suspended and deposited sediment. All rivers and streams must be managed for at least their current attribute state, where it is at or above the NBL.

This section addresses 'grading' the current state of rivers for each sediment attribute. This requires an assessment of existing monitoring data. Below is guidance on this field monitoring.[6]

## 3.1 Suspended fine sediment and visual clarity

The required visual clarity metric for site grading is the median of five years of at least monthly observations, that is at least 60 observations (table 8, NPS-FM 2020).

### Key considerations

1. Directly measure visual clarity.
2. For councils still measuring turbidity, convert turbidity records to visual clarity.
3. Calibrate measurements at site and by instrument.
4. Continually monitor and record.

[6] Although defining FMUs and associated monitoring sites for fine sediment attributes is a necessary first step, guidance on how to define/elect these sites is beyond the scope of this document.

### 3.1.1 Direct measurement of visual clarity

Manual direct measurement of visual clarity on a fixed interval basis is widely used by councils for state of environment monitoring. It is the simplest method of meeting NPS-FM monitoring requirements, including site grading.

Visual clarity is typically measured manually in situ, using a horizontal black-disc or, in turbid conditions, a clarity tube [7] following the procedures in the water quality National Environmental Monitoring Standard (NEMS-Water Quality 2019).

However, you can also measure visual clarity with a beam-c attenuation instrument in water samples taken to the laboratory (Davies-Colley 1988). Beam-c is best measured in the laboratory since the available instruments are not designed to work in rivers.

The water samples can be collected manually or by auto-sampler. They are measured using a green light beam transmissometer (NEMS-Water Quality 2019), where the green light LED is closely interconvertible with human measurements of visual clarity (Zaneveld and Pegau 2003).

Beam-c attenuation instruments measure the light beam attenuation coefficient (beam-c) as a total light attenuation by both absorption and scattering of dissolved and particulate materials at all angles in a unit of inverse metre.

To compare with in-situ visibility measurements, beam-c attenuation (c) can be converted to visual clarity (V), using the following relationship developed by Zaneveld and Pegau (2003). This accounts for attenuation by water itself, together with wavelength shift from the instrument wavelength (532 nm) to human eye (550 nm) sensitivity:

$$V = \frac{4.8}{0.9c + 0.81} \qquad \text{(Equation 1)}$$

### 3.1.2 Estimating visual clarity from turbidity

An alternative to direct measurement of visual clarity is to estimate it from turbidity data via a calibration relationship. This requires discrete manual measurements of turbidity with either a portable field instrument or a laboratory instrument applied to water samples from the field.

An alternative is to use an in-situ field instrument, recording turbidity data on a continuous basis at high frequency (eg, 5–15-minute intervals). Since this gives much greater resolution of visual clarity variations over time, it is preferable to manual turbidity measurements.

Indeed, since manual turbidity and visual clarity observations require similar effort, there is little point in not simply measuring visual clarity directly. However, where a site lacks any existing measurements of visual clarity, an existing record of monthly manual turbidity observations may be of use to initiate a site’s grading once calibrated to visual clarity.

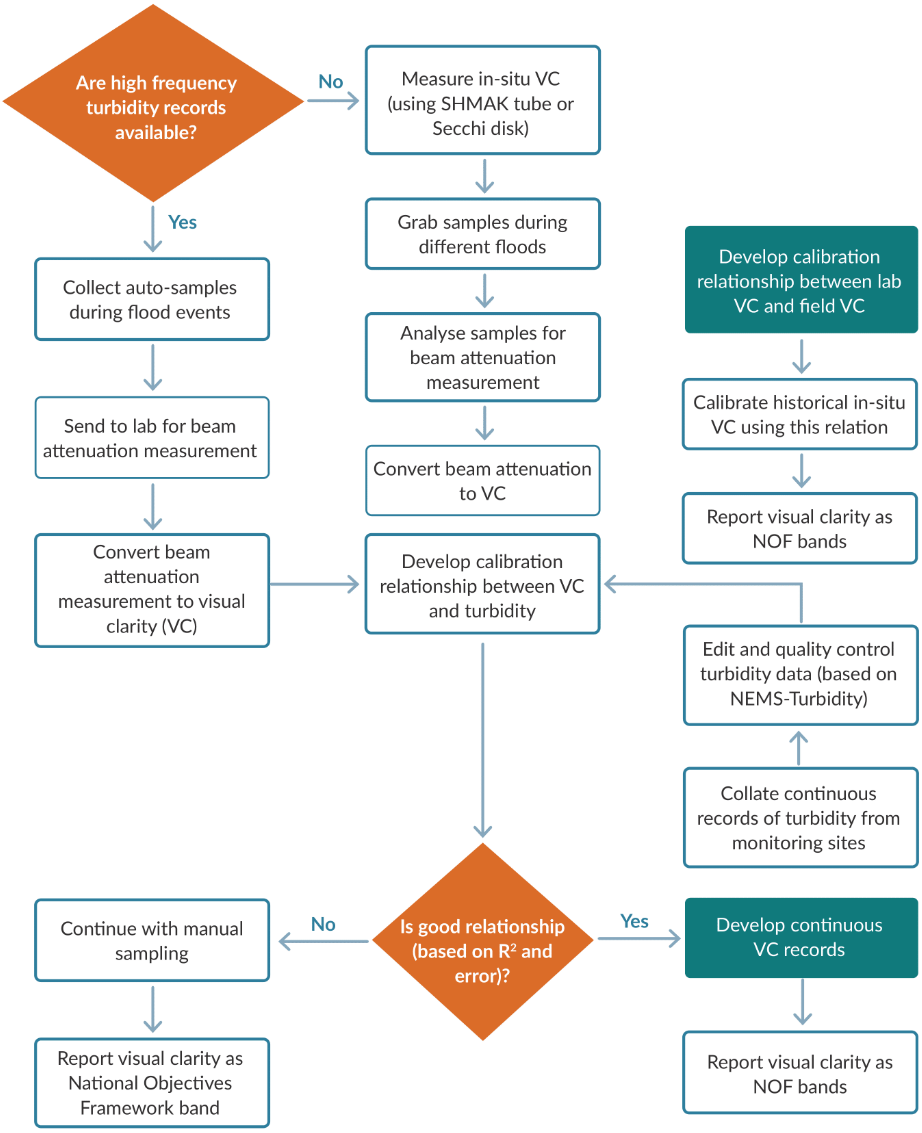
Generating a surrogate visual clarity record from an in-situ turbidity record requires a period of calibration through site-specific concurrent measurements of visual clarity and turbidity, as measured with the field instrument.

7 A clarity tube should only be used at the low end of visual clarity (<~0.5 m) – see NEMS (2019).

Site-specific and instrument-specific calibration is required because the relationship between turbidity and visual clarity varies with the characteristics of the suspended material (notably size and composition) and dissolved constituents (eg, tannins) in the water (Davies-Colley and Smith, 2001), and with the model, design and calibration of the turbidity sensor [8] (Bright et al, 2020; Davies-Colley et al, 2021).

Figure 3-1 shows a decision tree for deriving continuous records of visual clarity from turbidity records, and the choices for end-users applying the approach.

**Figure 3-1: Methodological flowchart for converting continuous turbidity records to visual clarity**




8 To ensure consistent comparisons between visual clarity and turbidity, turbidity records collected by sensors that comply with the ISO 7027 standard and follow Turbidity NEMS instructions for post-processing and quality control should be used against visual clarity measurements. The unit of these turbidity measurements is FNU, indicating alignment with ISO 7027. However, if historical turbidity measurements associated with alternative turbidity standards (eg, with NTU units) are available, these can also be used to develop proxy records of visual clarity, provided the calibration relationship is developed from turbidity devices meeting the same standard.

Based on the resources available (eg, auto-sampler, black or Secchi disks[9] or visual clarity tube[10]), samples can be taken from the monitoring site to the laboratory for analysis or visual clarity can be measured in situ. Collect all calibration and subsequent validation measurements/samples as close as practicable to the turbidity sensors.

Caution is needed in using turbidity as a surrogate for visual clarity because the median condition being targeted typically occurs during base flows. Under these conditions, the 'signal to noise' ratio for a turbidity sensor tends to be low, due to the influence of measurements close to the sensor's sensitivity and the confounding influence of particle properties and other constituents (eg, dissolved colour).

Sensor biofouling can also be problematic at base flows, particularly in streams that receive high levels of nutrients and light. Pay close attention to the uncertainty of the relationship (ie, the degree of data-scatter) with the median turbidity value, with careful editing of the raw turbidity record to remove any data corrupted by fouling. The water quality National Environmental Monitoring Standard document (NEMS-Water Quality 2019) provides guidance on editing raw turbidity data.

Still, an added benefit of using a continuously recording turbidity instrument as a visual clarity surrogate is that it can also be calibrated to provide a surrogate record of suspended sediment concentration (see Water Quality and Suspended Sediment NEMS documents). This gives a continuous record of the suspended sediment load when combined with a co-located water discharge record. In the NPS-FM context, an ongoing continuous sediment load record is of value in:

- refining relationships linking the sediment attributes to suspended load (section 4)
- calibrating and validating catchment sediment load predictive models (section 5)
- assessing the efficacy of erosion mitigation work on sediment load reduction targets (section 7)
- informing on the at-a-site relationship between deposited fine sediment cover and antecedent suspended load (if deposited fine sediment cover is also monitored at the visual quality monitoring site).

Some councils already operate turbidity instrument networks as proxies for suspended sediment concentration (SSC[11]) and suspended load determination. Rating these to visual clarity would take advantage of existing resources.

Councils can use a similar approach to convert existing discrete measurements of turbidity to visual clarity for sites with discrete monthly monitoring of turbidity but not visual clarity.

To avoid standardisation issues, you will need to develop a calibration relationship by compiling concurrent measurements of visual clarity and turbidity using the same turbidity

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9 The Secchi disk method takes vertical measurements through the water (Secchi depth is typically about 25% greater than black disc visibility).

10 For example, the Stream Health Monitoring and Assessment Kit (SHMAK) clarity tube.

11 We note here the distinction between the terms **suspended sediment concentration (SSC)** and **total suspended solids concentration (TSS).** This is broadly used in Aotearoa for water quality monitoring. Both terms represent the mass of suspended particulates in unit volume of water, but their laboratory analytical methods are different. SSC data are produced by analysing the complete volume of the original sample collected in the field. TSS data are produced by several methods, most of which analyse only a subsample of the original. See NEMS-Water Quality (2019).

measurement approach (ie, field measurement or laboratory analysis of water samples) and the same instrument standard (ideally, the same instrument) as used for monthly monitoring of turbidity.

Again, expect the calibration relationship to show scatter due to the variability of the fine sediment properties during runoff events. This will generate uncertainty in the derived hindcast estimates of visual clarity.

For this reason, continue calibration sampling until there is adequate definition of the relationship around the median values and the standard error on the estimated visual clarity. The number of samples required will be site-specific and depend on the extent of scatter in the relationship between visual clarity and turbidity.

Moving forward, in any discrete manual monitoring at these sites, include direct measurement of visual clarity, in preference to discrete estimates derived from turbidity.

## 3.2 Deposited fine sediment cover

### Key considerations

1. Use the SAM2 method.
2. Ensure monitoring team are well trained.
3. Determine if the river or stream is naturally hard bottomed.

The grading of deposited fine sediment cover applies to wadeable (and naturally hard-bottomed) rivers and streams, with the metric of the percentage cover of the visible streambed.

Table 16 (appendix 2B) of the NPS-FM (appendix A here) requires measurements to be made using stream assessment method 2 (SAM2), in line with guidance in Clapcott et al (2011). SAM2 is an in-stream visual assessment of the surface area of the streambed covered with fine sediment. It involves making a minimum of 20 visual cover estimates in run habitat.

Under the NPS-FM, current attribute state is determined from the median of 60 measurements over five years of monthly monitoring (or a longer period where flow conditions only permit seasonal monitoring).

This methodology is well specified by Clapcott et al, but can be manually intensive and is vulnerable to operator bias unless monitoring staff are well trained (Basher et al, 2020b). With each SAM2 survey, note also the dominant texture of the fine sediment cover (ie, whether sand or mud). This can be assessed visually through a bed-viewing device (eg, bathyscope).

Potential future methods of assessing fine deposited sediment cover, especially for large non-wadeable rivers, include remote-sensing approaches using classified imagery from piloted or remote-controlled air or water craft (eg, Niroumand-Jadidi et al, 2019). Currently, these methods are still in the research domain and require some ground-truth validation.

Councils can use a combination of field investigation, historical research and statistical study/GIS classification to determine whether a currently soft-bottomed wadeable stream would be a hard-bottomed stream under natural conditions.

The field investigation would include site assessments (eg, digging through the soft sediment to check for underlying hard substrate) and comparisons with adjacent/nearby reaches in similar geomorphic settings.

The historical research would assess the history of the stream and catchment, including talking to iwi, to see if there is any knowledge of it being hard bottomed in the past and to record events (eg, forest removal) that may have forced a change to soft-bottomed.

The GIS classification would attempt to predict the natural bottom state based on reach characteristics. A study by Haddadchi et al (2018) used reach characteristics to predict river bed substrates.

This and similar studies could be used as a guide to direct the more intensive field and historical research effort (ie, by focusing investigation on reaches where the predicted substrate is hard but the observed substrate is soft).

As a starting point, councils may wish to use the GIS layer produced by Haddadchi et al (2018), to check field condition against the model prediction. If these differ, then it warrants further investigation and local field inspections. To access this tool, see the NZ River Maps website.

# 4 Linking TAS to sediment load

This section has guidance on estimating how much to reduce the catchment sediment load to achieve TAS for visual clarity and deposited fine sediment cover, when monitoring has shown that the current state does not meet the target state. Councils should quantify the load reduction target needed to meet the TAS before setting resource use limits to achieve it.

See NPS-FM, subpart 2.

In this section we treat visual clarity and fine sediment cover separately, mainly because more is known about the sediment load's link to visual clarity than to fine sediment cover.

## 4.1 Linking visual clarity to sediment load

### Key considerations

1. Quantify load reduction targets required.
2. Use the 'simple' approach (proportional reduction in a stream) on all - and then use
3. the 'complicated' approach if a substantial proportional sediment reduction is needed.
4. Use models to map and mitigate sediment sources.
5. Monitor mitigation progress.

### 4.1.1 Overview

This section presents both a simple approach and a more complicated approach for estimating the change in suspended sediment load required to achieve the visual clarity TAS.

With visual clarity, the focus is on the suspended sediment load. This is because visual clarity is inversely related to particle size, and so tends to be dominated by clay and fine-silt grade sediment that is transported in suspension.

A key point is that median visual clarity (the attribute adopted for the NPS-FM) typically occurs during base-flows or well down the recessions of storm runoff events. Therefore, in theory, the most expedient way to improve the median clarity should be to mitigate sediment sources that are active during those flow conditions.

However, identifying those sources is difficult, whether by tracing in the field or by numerical simulation in dynamic sediment erosion and routing models, so is still in the research domain.

The simple approach (4.1.2 below), uses a set of simplifying assumptions. It assumes that suspended sediment load reductions effected at erosion sites are proportionally reduced across the hydrograph. For example, a 50 per cent reduction in the mean annual suspended sediment load[12] from a catchment is manifest as a 50 per cent reduction in average sediment concentration across all discharges and a 50 per cent reduction in the median concentration.

When combined with a relationship between sediment concentration and visual clarity, this enables an estimate of the mean annual load reduction required to induce a given change in median visual clarity.

The complicated approach involves dynamic modelling (see 4.1.3). It avoids these assumptions and can also quantify the load reduction required of time-dependent sources. It can provide greater resolution and confidence, but at substantial extra effort and cost.

### 4.1.2 Simple approach

This approach is detailed in Hicks et al (2019a). It simplifies further the approach developed by Hicks et al (2016), and published by Dymond et al (2017).

The key feature is that it determines only the proportional reduction in stream suspended load required to achieve the TAS. Explicit values of the current and target sediment loads and the reduction in load are not required. The only data required for the monitoring site are:

- the current median visual clarity, $V_{50}$
- the target median visual clarity, $V_{t50}$ , and
- the exponent in the relationship between visual clarity and suspended sediment concentration.

The advantage of working with proportional load reductions is that information on sediment sources (eg, via tracing methods or an uncalibrated distributed erosion model) and erosion mitigation plans (eg, a change in catchment land cover) need only be proportional as well to make the required load reduction at the monitoring site (see section 7).

The essential elements are:

- We expect, based on the findings of Hicks et al (2016) that at a site, visual clarity (V: m) will fit a power-law function of suspended sediment concentration (C: mg/l):

  $$V = gC^{d} \qquad \text{(Equation 2)}$$

  where *d* and *g* are site-specific empirically derived coefficients. Thus, the median visual clarity ($V_{50}$) is related to the median concentration ($C_{50}$) as $V_{50} = gC_{50}^{d}$.

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12 Mean annual suspended load is defined as the mass of suspended sediment discharged over a multi-year period divided by the duration of that period, with units t/y. Typically, monitoring periods of a decade or more are required to provide a reasonably stationary estimate of measured mean annual load. Empirical steady state models (section 5) aspire to predict true long-term mean annual load. The mean annual suspended sediment load is considered the most appropriate suspended load statistic to link with because:

- a long-term (multi-year to decadal) central statistic is needed to smooth out the substantial variability in loads observed over shorter timeframes (eg, events, annually) due to hydrological variability (Davies-Colley and Smith 2001)
- the mean annual load is the conventional statistic output from suspended sediment monitoring programmes and from steady state and dynamic models that predict sediment loads (section 5), and
- median visual clarity can be analytically linked to the mean annual load (eg, as developed here in subsection 4.1.2).

- Catchment **mean annual suspended sediment load** can be derived using a current sediment rating curve and flow frequency distribution. The sediment rating curve is usually expressed in the form $C = aQ^b$, where $a$ and $b$ are site-specific coefficients and Q is water discharge (l/s), thus the existing catchment sediment load (L: mg/s) is calculated as:

$$L = \sum p_i\, aQ_i^{b+1} = aQ^* \qquad \text{(Equation 3)}$$

where $p_i$ are the proportions of time that discharges are within each discharge band ($Q_i$) and $Q^* = \sum p_i\, Q_i^{b+1}$.

Q* may be regarded as a function of the catchment hydrology and rating curve slope ($b$), which are both assumed not to change if the sediment load is reduced (ie, only the rating curve offset ($a$) changes).

- From the above, we get $a = (C_{50}/Q_{50}^b)$ and $C_{50} = (V_{50}/g)^{1/d}$, thus:

$$L = aQ^* = Q^*(V_{50}/g)^{1/d} / Q_{50}^b \qquad \text{(Equation 4)}$$

- If we let $L_t$ be the target sediment load and $V_{t50}$ is the defined limit for $V_{50}$, then:

$$L_t = Q^*(V_{t50}/g)^{1/d} / Q_{50}^b \qquad \text{(Equation 5)}$$

- Finally, the load reduction factor (LRF) may be expressed as:

$$LRF = (L - L_t)/L = 1 - L_t/L = 1 - (V_{t50}/V_{50})^{1/d} \qquad \text{(Equation 6)}$$

In other words, the proportional reduction (if any) in catchment sediment load (R) required to increase visual clarity to the TAS is a simple function of the ratio of current median clarity and target median clarity, where the exponent $d$ can be derived, by preference, from data collected at the monitoring site or can be assumed to take regional or national average values[13] in the absence of an adequate dataset for the site. Note that this proportional approach does not require an explicit value for the coefficient $g$ in equation 2.

For example, a monitoring site on a river reach within Suspended Sediment Class 1 is graded with a 5-year median visual clarity ($V_{50}$) of 0.8 m (ie, band D in table 8, NPS-FM 2020, below the NBL of 1.34 m), and the exponent ($d$) in the observed relationship between visual clarity and suspended sediment concentration is -0.76. Equation 6 shows that the mean annual suspended load of the catchment needs to be reduced by 49 per cent (ie, to 51 per cent of its current load) to achieve the national bottom line median visual clarity ($V_{t50}$) of 1.34 m.

The simplification represented by equation 6 develops because the terms Q*, $g$, and $b$ at the site of interest are assumed constant, so they cancel out for the ratio of the existing and target states. However, as discussed in Hicks et al (2016), after catchment changes to mitigate erosion, the Q* function may change due to either a change in the sediment rating curve slope b, or a land cover-driven change in runoff and the flow frequency distribution.

As shown by Warrick (2015), while reductions in catchment sediment supply do often simply result in a 'downshifted' sediment rating plotted on a log-log-curve, concurrent changes in water discharge regime can influence this vertical shift, and/or pivot the rating curve.

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[13] Hicks et al (2019) derived a national average over 77 sites of d = -0.76 ± 0.13, where the uncertainty is the standard deviation.

Moreover, changes in the spatial distributions of sediment sources and runoff over a catchment can also drive change in both coefficients *a* and *b* of the sediment rating. It that is expected or observed, then it may require a more complicated approach to estimating sediment load reduction.

This simple approach is used by Hicks et al (2019a) to model the proportional load reductions required to achieve sediment targets across Aotearoa. It directly informed the development of the NPS-FM (2020) sediment provisions.

These proportional load reductions were then used to model availability of the proposed sediment limits based on turbidity data (Neverman et al, 2019). However, due to the inconsistency of turbidity values, wherever possible we recommend using visual clarity values to determine sediment limits.

### 4.1.3 Complicated approach

We recommend this approach where the simple approach indicates a substantial proportional reduction in catchment mean annual suspended load (and hence a greater need for precision of estimates) – particularly where the mitigation options involve changes in land cover, or are not uniform over the catchment, rendering invalid the stationarity[14] assumptions underpinning the simple approach.

For example, mitigation might focus on bank erosion, and the measures might include channel modifications that alter flow hydraulics (eg, riparian planting) or that alter catchment drainage (eg, creating flood storage in wetlands or artificial retention basins). Both of these can alter runoff rates and so alter the flow distribution and sediment rating.

Ideally, this should take the form of a spatially distributed, dynamic, catchment erosion and hydrological model that routes runoff and sediment past the monitoring site, producing a simulated record of SSC which can then be converted to visual clarity. In this case, there is no explicit determination of the load reduction factor since this will need to be determined iteratively by 'applying' erosion mitigation into the model.

A more complex model does not necessarily deliver greater accuracy or improve confidence (Schoups et al, 2008; Orth et al, 2015). More complex models require more parameters and data, which typically requires extensive calibration to fit the models to the available data.

Therefore, it is important to demonstrate model performance compared to measured data including in-stream measurements (eg, SSC, visual clarity), and measurements from erosion sources (relative contribution of sediment sources identified by techniques listed in section 6).

In essence, it requires merging this step with those in section 5 and section 6, using a catchment model to both map and mitigate sediment sources on an iterative, trial-and-error basis until the simulations predict a median visual clarity at or above the desired target state.

The most useful dynamic models should also provide output that identifies sediment sources that are active while suspended sediment concentrations and visual clarities are around their median values.

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[14] Stationarity is a situation where the sediment load may change over time but the relationship between it and the factors affecting it remains constant.

For potential models, see section 5 and table 5-1. Given the substantial resource investment these require, users will need to undertake further due diligence before selecting the best model for their catchments.

### 4.1.4 Guideline for choosing visual clarity approach

We recommend using the simple approach in all cases, then progressing to a more complicated dynamic model. Use the best available option if the stationarity assumptions of the simple approach (ie, that the parameters defining the flow frequency distribution and the relationships of sediment concentration with visual clarity and discharge do not change after mitigation) are either expected or observed to be violated after mitigation.

To assess these stationarity assumptions, councils must:

- assess if the mitigation options under consideration involve changes in land cover or are not uniform over the catchment in their location or erosion type (eg, focus on mitigating bank erosion), and
- while mitigation is in progress, collect data to monitor for any significant change in the parameters defining the flow frequency distribution and the relationships of suspended sediment concentration with visual clarity and discharge.[15]

## 4.2 Linking deposited sediment to sediment load

#### Key considerations

1. Sediment phasing is important.
2. Size grading required at the site (mud or sand).
3. Select appropriate mitigation.
4. Be alert to transient signals.
5. Record extreme hydrological events.

### 4.2.1 Overview and review

Deposited fine sediment spans the particle size range from clay to sand (0.001–2 mm) and occurs during floods. Linking deposited sediment with the mean annual suspended load (which is dominated by flood runoff) is appropriate for deposits of mud (ie, clay and silt, finer than 0.063 mm), but sand deposits (0.063-2 mm) may occur from either the suspended or bed load.

However, since bedload is rarely measured, but often estimated as equivalent to a proportion of the suspended load (Hicks et al, 2004), it is expedient to also link deposited sand cover to the mean annual suspended load.

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[15] This monitoring should be adequately covered by the state and trend monitoring specified in section 8, and by the discharge record at the site.

Even though it is intuitive that fine sediment cover on streambeds should be positively related to catchment sediment load (ie, a reduced load should result in less deposited fine sediment), there is scant information on at-a-site relationships (in contrast with those available for visual clarity).

Datasets that have been used to explore the factors influencing deposited fine sediment cover come from many sites across Aotearoa. However, most of these sites have only one to several records. The relationships derived relate more to factors controlling spatial variability than to those influencing temporal changes at-a-site and are likely blurred by a high sampling error on site-representative values.

Predictive models generally show either only a weak correlation of deposited fine sediment cover with specific mean annual sediment load[16] or even an inverse correlation (eg, Hicks et al, 2016). This conflicts with the notion of managing deposited fine sediment cover by reducing upstream catchment loads.

Hicks et al (2019a) developed a conceptual, physically based, model which suggested that deposited fine sediment cover should be also influenced by the phasing of sediment delivery during floods and site local hydraulic factors, with streambed deposition enhanced on flood recessions when sediment delivery from upstream remains high, while sediment transport capacity wanes rapidly over time and downstream (such as at slope breaks).

In the only comprehensive space-time assessment of deposited fine sediment cover in an Aotearoa catchment, Basher et al (2020a) monitored deposited fine sediment cover at 30 sites across the Motueka catchment (Tasman District) over six years. The fine sediment was mainly sand.

They found that large floods and sediment pulses from forest harvesting induced only small, localised and transient changes in deposited fine sediment cover. This reflects the diffusive (attenuating) nature by which sand 'slugs'[17] move downstream under bedload transport (as against the more advective transport of mud-grade suspended sediment).

Basher et al (2020a) found that local delivery of sediment and channel morphology (and so hydraulics) were more important controls on catchment-wide deposited fine sediment cover patterns than upstream annual sediment load. However, at some sites they did find strong linear correlations between deposited fine sediment cover and antecedent suspended load (accumulated over less than two years).

These studies indicate that assessments of load reduction requirements should consider the size grading of the deposited sediment at the monitoring site (ie, whether mud or sand). If it is predominantly sand, then reductions in the relatively local sand sources may yield a better impact on in-stream deposited fine sediment cover than catchment-wide reductions in the suspended load over all sediment size fractions.

If the deposited fine sediment cover is predominantly mud, a broader treatment area will likely be appropriate. However, there should be particular emphasis on erosion sites delivering sediment to the monitoring site on flood recessions.

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[16] Specific mean annual load is the mean annual load divided by catchment area.

[17] 'Slug' means the slow migration downstream of a discrete volume of sand that is released into a stream channel after a large erosion event.

Also, if the deposited fine sediment cover is mainly sand, be aware of any recent events that may have initiated a transient, migrating sand slug. This could be misinterpreted as a permanent change or trend over the NPS-FM five-year monitoring cycle.

### 4.2.2 Guideline for fine sediment cover

Because there is scant research on the relationship between deposited fine sediment cover and sediment load at-a-site, we recommend:

- assuming initially a linear response between load change and deposited fine sediment cover, as observed by Basher et al (2020a), until better informed by the monthly deposited fine sediment cover state monitoring (as described in section 3), particularly if accompanied by suspended load monitoring, and
- aligning the spatial extent of mitigation with the grainsize of deposited sediment (ie, whether mud or sand), but
- being alert for transient signals in deposited fine sediment cover stemming from extreme hydrological events.

For example, if comparison of the current state (median deposited fine sediment cover) and desired TAS indicates that cover needs to be reduced by 50 per cent, and visual assessment shows that the deposited fine sediment cover is dominated by sand, then, providing the cause is not a transient event, mitigation efforts should aim for a 50 per cent reduction in sediment delivery from local sand-rich sources.

The first step is to identify sources of sediment for different particle sizes. (See section 6 for methods to identify sediment sources; see example studies by Haddadchi et al (2015) and Vale et al (2020) that discriminated sediment sources by particle size using sediment fingerprinting techniques.)

Then, apply mitigation strategies for the dominant sand-rich source. If the deposited fine sediment cover is mainly mud, then mitigation efforts should aim for a 50 per cent reduction in the total suspended load.

However, focus on sources that deliver sediment to the site late in the hydrograph (eg, from the catchment headwaters, from where the travel time is long, or from eroding banks, which often tend to collapse on recessions as saturated banks are exposed).

# 5 Estimating catchment suspended sediment load

After linking fine sediment attributes to catchment sediment loads, the next step is to quantify the catchment suspended sediment load.

This section describes the modelling and monitoring approaches to do this.

See NPS-FM, subpart 2.

## Key considerations

1. Modelling and monitoring are complementary and not alternatives.
2. Models should be tested against monitored data that is representative of the catchment.
3. Long-term monitoring is required before and after an activity.

In the NPS-FM context, estimates of a river's suspended sediment load are useful for several purposes in the NPS-FM sediment workflow, including helping to:

- quantify the magnitude of sediment load reduction required to meet sediment attribute targets
- formulate and map catchment sediment budgets to identify where to focus mitigation
- improve and validate the accuracy of predictive models
- inform on the effectiveness of sediment management strategies.

Knowledge of sediment load also informs catchment management to the benefit of sediment issues in downstream coastal systems, including estuarine ecosystem health, nearshore fisheries, and sand supplies to eroding coasts.

Monitoring and modelling should be seen as complementary approaches, rather than alternatives, when quantifying a catchment's suspended sediment load. For example, in small well-monitored catchments, it may be possible to estimate sediment loads from monitored data. In larger catchments, however, a high level of monitoring may not be economically or logistically feasible, so modelling becomes necessary to provide adequate spatial coverage.

Indeed, under the NPS-FM (2020), modelling can provide information in the absence of complete and scientifically robust data (ie, monitored data). The proviso is that local authorities "take all practicable steps to reduce uncertainty (such as through improvements to monitoring or the validation of models used)" (NPS-FM, clause 1.6, Best information).

Where there is uncertainty, councils must set resource use limits based on a precautionary approach that prioritises Te Mana o te Wai (the health of the waterbody) using "safety first" assumptions.

The models informing these assumptions should be tested against monitored data that represents the full set of catchment characteristics and physical conditions for which the model will be run. This ensures that the model can be transferred over time and space.

Models are also required for scenario testing to aid in the planning and implementation of strategies to attain or maintain the fine sediment objectives; for example, to locate hotspots requiring erosion control, to assess the best erosion control options and their likely downstream impacts, and to determine where further monitoring may be required to fulfil data requirements.

Table 5-1 compares the two approaches based on the following criteria:

- Associated costs and timeframes to achieve the outcomes of the modelling or monitoring techniques.
- Ability to quantify the effects of changes in resource use on catchment sediment load.
- Capability of the monitoring and modelling techniques to forecast and evaluate the impact of sediment control strategies.
- Transferability of the results derived from modelling or monitoring techniques to visual clarity and its relevant NOF attribute bands.
- Ability to identify and locate sediment sources and erosion processes.
- Ability to quantifying the impact of bank erosion on suspended sediment load and sediment attribute states.
- Ease of conducting temporal trend analysis.
- Ability to evaluate uncertainty associated with the results.

**Table 5-1: Comparing sediment load modelling and monitoring techniques across a range of criteria**

| Modelling sediment load | Monitoring sediment load |
|---|---|
| **Budget and timeframe** | |
| Modelling is less expensive than monitoring. For pre-existing models, the only time required is to collate the input data and layer, and then calibrate the model based on available sediment load dataset. | Setting up a monitoring site, servicing the instruments, collecting samples during and after floods (to make a direct estimate or to calibrate surrogate sensors such as turbidity), and editing and quality assurance of the sensor records are labour intensive and expensive. |
| **Land use change scenario-based assessments** | |
| Most of the available models can be used to evaluate changes in resource use such as land use change. This can be done by adjusting model input layers and databases. | Monitoring can evaluate the impact of land use on sediment load and sediment attributes by setting up a network of monitoring sites in sub-catchments with different land uses but similar settings for other catchment characteristics including terrain, soil type, climate and hydrology. |
| **Predicting or evaluating the impact of mitigation** | |
| If information is available about sediment trapping efficiency and effectiveness of the ESC practices, the modelling techniques can be adjusted to predict their mitigation impacts (see section 7).<br><br>Different ESC practices can be tested at different parts of the catchment, to find the best location and the most appropriate option based on the costs and mitigation impact. | Monitoring can only validate the effectiveness of erosion control strategies that are in place. To do this, monitoring is required either long term before, during and after the implementation of the strategies, or using paired catchments with and without the strategies in place. |

| Modelling sediment load | Monitoring sediment load |
|---|---|
| **Transferability of the results to visual clarity** | |
| Sediment load results from the models can be converted to discharge-weighted suspended sediment concentration. Visual clarity can then be estimated by using the calibration relationship between SSC and visual clarity. | As with modelling, as long as a calibration relationship between sediment concentration and visual clarity exists for the specific study site, visual clarity values can be derived from sediment concentrations (or directly from turbidity if that is monitored as a SSC-surrogate). |
| **Identifying sediment source hotspots** | |
| When set up with adequate granularity, most of the available modelling approaches can provide sediment source maps on which hotspots will be identifiable for the period of the load estimate (ie, sediment source maps for the mean annual load for models with long-term estimates and high temporal resolution sediment source maps for intra-event load modelling). | Monitoring can inform about erosion sources within a catchment if there is a network of nested monitoring sites across the catchment.<br><br>But the spatial resolution is limited to the areas of the monitored sub-catchments, and there is a practical limit to the number of monitoring sites.<br><br>This means that a nested monitoring network cannot generally pinpoint the contributions from discrete sources or from different land use, erosion type or geological units. |
| **Quantifying the impact of bank erosion on sediment and attributes** | |
| Based on the type of the model, estimates of bank erosion contribution to suspended sediment load can be included explicitly or implicitly. Models with explicit estimation of bank erosion assess the physical processes in operation.<br><br>Other models implicitly include the net sediment delivery from eroding banks (since bank erosion contributions will be captured at the calibration sites along with sediment from other erosion sources, such as hill-slope erosion, landslides and gully erosion). But isolating the bank erosion contribution is either not possible or requires model enhancement. | The contribution of bank erosion (net of any riparian deposition) to the monitored suspended sediment load can only be determined by differencing loads measured upstream and downstream of reaches with actively eroding banks and no significant lateral sources. |
| **Trend analysis** | |
| Steady state models with long-term derived estimates of sediment attributes do not provide the data required for temporal trend analysis.<br><br>Dynamic models can generate time-series output which can be analysed for direction and rate of change, for example to predict the effectiveness of mitigation plans. | Continuous records of suspended sediment concentration and load can be used to monitor progress towards achieving load reduction targets.<br><br>Continuous records of sediment load and concentration can be used in trend analyses to identify degradation or improvement in the fine sediment attributes. See section 8 for further details. |
| **Uncertainty of load estimates** | |
| Higher uncertainty in estimating sediment loads at-a-site compared with monitoring, but models have the advantage of providing spatially distributed results.<br><br>The uncertainty will also transfer to sediment related attributes. This is lessened if the models are calibrated or validated off sediment load monitoring at key sites. | Since sediment load monitoring data comes from direct measurement of the SSC in rivers or calibration of surrogate measurements (eg, turbidity), monitoring has lower uncertainty than modelling.<br><br>Monitoring is, however, spatially limited by cost and resources. |

# 5.1 Modelling suspended sediment load

In this section, we give general guidance on model choice, discuss the types of sediment models available, and give examples of these and their use in Aotearoa.

We also discuss sources of uncertainty for sediment modelling.

We do not make recommendations on which models to apply. The choice will vary on local data availability, time and budget constraints and the purpose of the modelling.

## Key considerations

1. Seven criteria guide which model to select:
    a) objective and purpose
    b) temporal and spatial scale
    c) data requirements and availability
    d) model suitability and accessibility
    e) usability
    f) set-up and running time
    g) track record and support.
2. Apply a sediment load reduction factor to mitigations, erosion and sediment controls to incorporate into models.
3. To reduce model uncertainty: improve the current level of sediment and as needed flow monitoring, so that future modelling can draw on these data for model calibration and testing.

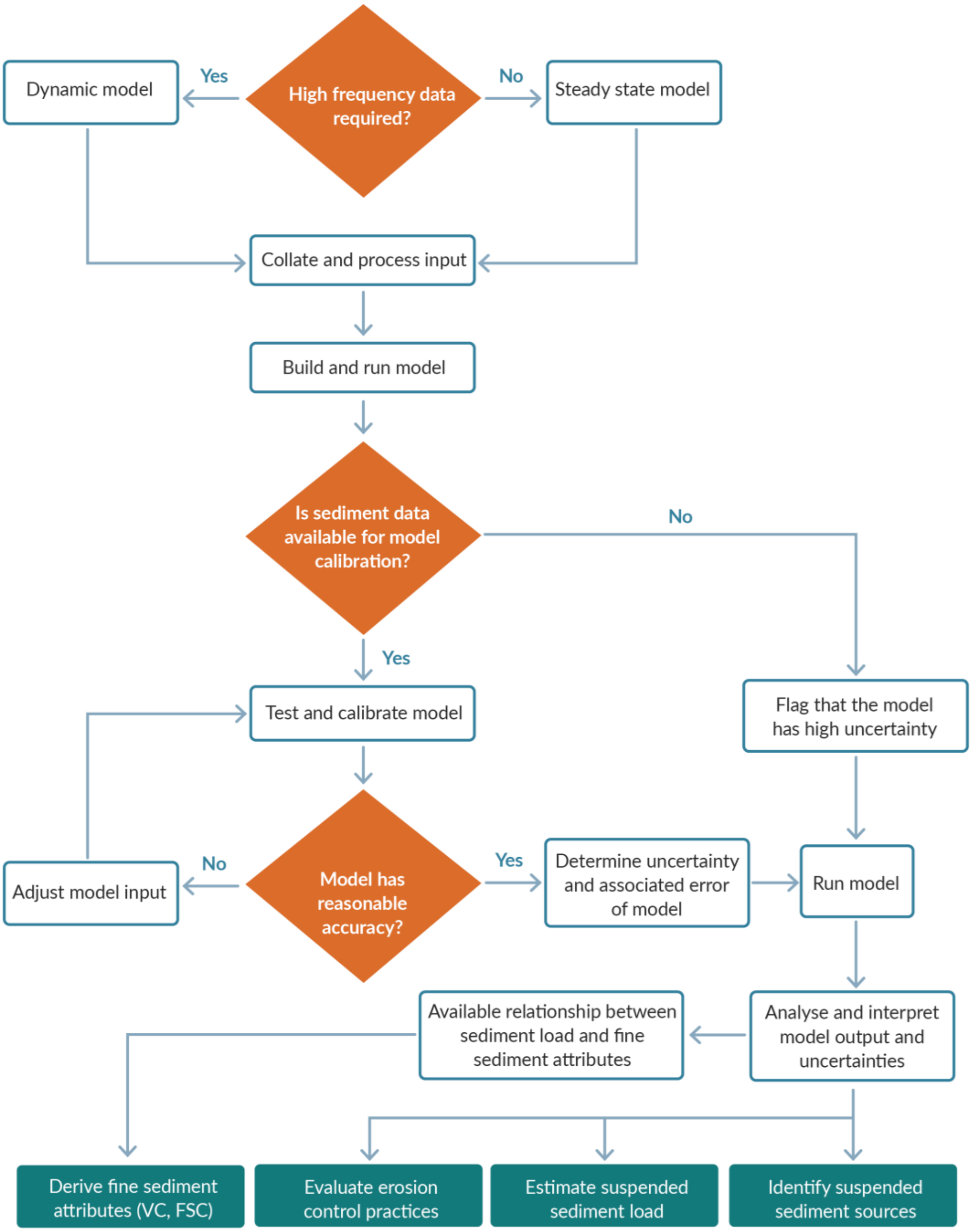
Catchment sediment loads may need to be estimated from modelling, particularly where monitoring data are limited or lacking.

Models can provide sediment load estimates at a variety of spatial and temporal scales. You can also use them to derive other metrics for sediment (eg, visual clarity) provided that suitable relationships between the sediment and these metrics can be established (see section 4).

Models can also be used to map and identify catchment sediment sources (see 6.3) and evaluate the impact of different ESC practices on downstream sediment loads and fine sediment attributes (see 7.4).

Figure 5-1 shows the workflow of the key steps for estimating suspended sediment load, estimating fine sediment attributes, identifying sediment sources, and evaluating erosion sediment control practices using different modelling techniques.

**Figure 5-1: Steps for estimating suspended sediment load, identifying sediment sources and evaluating erosion control practices using different modelling techniques**




High-frequency dynamic modelling is recommended for large, heterogenous catchments where simple assessments indicate the need for substantial erosion mitigation efforts.

### 5.1.1 Model type and choice

After determining the objective, and which questions require answering, the choice of any environmental model is guided by the following principles:

1. **Model purpose**. Questions include:
    - What outputs are required?
    - How will we report the outputs (eg, charts, tables, maps)?
    - How will we use the outputs (eg, research, catchment planning, policy development, rule setting)?

2. **Spatial and temporal scale.** Questions include:
   - Will we apply the model to a small sub-catchment or at a regional scale?
   - Are time-series required, and if so, at what timestep?
   - Are the scales appropriate to the processes modelled?

3. **Data requirements and availability.** Questions include:
   - What data are required to run, calibrate and test the model? Is that data available at the model spatial and temporal scale?
   - If the data needs to be up- or down-scaled, what approaches will we use, and what are the possible impacts on model uncertainty?

A pre-requisite for sediment model calibration and testing is that there is sufficient data on sediment concentration and concurrent flow.

These would preferably be from as many sites as possible that represent catchment characteristics, to allow the estimation of measured sediment loads (see 5.2).

The data for model calibration should be of long enough to include the full range of flow conditions to establish a reliable relationship between flow and sediment concentration.

Sampling should also include high-flow events since these are typically associated with high erosion events and therefore sediment loads.

4. **Model suitability.** Questions include:
   - Which models can fulfil the model purpose and what is their availability and, for proprietary models, cost?
   - Will different models be required for different aspects of the modelling, and if so, how well can these models be coupled?
   - Does the model have the appropriate level of complexity to achieve its purpose with available data?
   - Has the model been developed for the model purpose and, if not, how transferable is the model?
   - Can the model be adapted?

5. **Model assumptions.** These vary by model and are generally related to the way the model represents physical processes. Describe and report the assumptions for consideration. Model assumptions are discussed in subsection 5.1.5, in relation to model uncertainty.

6. **User group and model usability.** Questions include:
   - Who is going to use the model, and what is their skill level?
   - Will the model be publicly available, or is it intended to be run by an expert?
   - Does the model have a user-friendly interface?
   - Are user support and training available?

7. **Model set-up and running the model.** Questions include:
   - What needs to be done to setup and run the model?
   - Will the model require modification, and if so, is the source code available?

- If scenario modelling is required, can scenarios be easily developed and run?
- What computational power is required?
- How long are the run times?

8. **Track record and support.** Questions include:
   - What is the model provenance? Is it well known or trusted?
   - What is the level of performance and uncertainty shown by the model in previous applications?
   - What is the model longevity and has the model been maintained?
   - Is it easy to update the model with new versions?
9. **Cost.** How much will the model cost to use, including licensing of the model itself and any dependant software (eg, GIS), data collection and management, set-up and running, consultancy, and, if required, hardware upgrades? What will version updates cost?

These questions can help the user to select an appropriate model based on different circumstances. For example, a national or regional steady-state model will have different data and computational needs from a dynamic model applied at the farm scale, or for a single catchment or sub-catchment.

Model choice can be a compromise where the objective is to provide the best possible model outputs, using the data available, while minimising model uncertainties. It is not uncommon that one model, on its own, cannot fulfil all the criteria for a model application.

This may mean that models need to be coupled (eg, catchment accounting models coupled to river contaminant transport models). Where two or more models are coupled, ensure that they are both testable and have similar levels of complexity and spatial and temporal scales (Tscheikner-Gratl et al, 2019).

### 5.1.2 Sediment load models in Aotearoa

There are two broad classes of models for simulating spatially distributed suspended sediment loads. Most of them deal with multiple water quality contaminants, not just sediment:

- **steady-state models** that predict the long-term average water quality state (eg, mean annual suspended load)
- **dynamic, or time-stepping, models** where an estimate of the water quality state is made for every time-step in the model (eg, daily or hourly SSC, visual clarity).

**Steady-state models** are generally used for screening to give a broad picture of long-term average state or contaminant delivery to receiving environments. They tend to have fewer data needs, simpler model builds, and faster run times. This makes them suitable for large-scale application (eg, national, regional or catchment) and scenario modelling.

Ideally, steady-state models are calibrated and tested using data records long enough to include the full range of conditions driving erosion and sediment transport, to allow calculation of average loads.

For example, SedNetNZ (Dymond et al, 2016) represents the contribution of episodic rainfall-triggered landslides to the long-term average annual sediment load. It does this using data on landslide erosion measured from aerial imagery over decades (up to 70 years), reflecting the impacts of multiple storm events.

The model averages this landslide-erosion contribution over time to produce average annual sediment loads from sources that are infrequent (eg, rainfall-initiated landslides) and frequent (eg, surface, bank, earthflow, gully) (Hugh Smith, personal communication, April 2022).

In contrast, **dynamic models** are more traditionally suited to problems where the timing of inputs to receiving water bodies is critical. However, they require more data, longer set-up times and more computational effort. This means their use is usually restricted to single catchments or sub-catchments.

Models can also be classified by their spatial and temporal resolution, and:

- the extent to which they are spatially distributed (eg, lumped by catchment or sub-catchment, represented by hydrological response units (HRUs) with similar combinations of land cover and topography, or gridded)
- their complexity (ie, the processes included) and
- their calculation methods (ie, statistical/empirical models versus process/physically based models).

Table 5-2 and table 5-3 summarise steady-state and dynamic water quality models that have been used to model sediment generation from land surfaces, and the transport and loads of these sediments in river networks in Aotearoa.

This summary is based on a stocktake conducted as part of an assessment of the requirements to create an interoperable model framework (Elliott et al, 2014). Steady-state models are listed in table 5-2 and dynamic models in table 5-3. For details such as model outputs, data requirements, model scale and modelling approaches, see the Interoperable Models WIKI.

Since the stocktake, NIWA has developed two further steady-state models (table 5-2):

- the Waikato Auckland Northland Sediment Yield estimator (WANSY; Haddadchi and Hicks, 2016)
- New Zealand Sediment Yield estimator (NZSYE; Hicks et al, 2019b) which updated an earlier estimation of national sediment loads (Hicks et al, 2011).

Dynamic models in table 5-3, used for modelling sediment in catchment planning, include:

- eWater's Source, applied in Bay of Plenty and Greater Wellington
- the Loading Simulation Program C++ (LSPC) model, customised for the Auckland Freshwater Management Tool (Bambic and Riverson, 2017; Grant et al, 2018).

**Table 5-2: Steady-state models used in Aotearoa to estimate sediment load generation and transport**

| Model | Developer | URL | Aotearoa context | Key references | Status and availability | Comments |
|---|---|---|---|---|---|---|
| NZEEM (NZ Empirical Erosion Model) | Landcare Research/ Manaaki Whenua | - | Developed for Aotearoa and applied nationally | Dymond et al (2010) | Closed source, not publicly available<br>Outputs available from the LRIS web-portal. | 15 m resolution grid-based model |
| SedNetNZ (Sediment budgets for river networks) | Landcare Research/ Manaaki Whenua | http://tools.envirolink.govt.nz/dss/sednet/ | Adapted for Aotearoa from the Australian SedNet model<br>Applied to catchments in Northland, Hawkes Bay, Waikato, Bay of Plenty and Manawatu | Dymond et al (2016)<br>Basher et al (2020b)<br>Vale et al (2021) | Closed source, not publicly available | 15 m resolution grid-based model<br>Separate calculation of sediment loads due to surface erosion, gully erosion, bank erosion, landslides and earthflows<br>Model is actively maintained with several updates since initial development |
| SPARROW (SPatial Regional Regression On Watershed attributes) | United States Geological Service (USGS) | http://water.usgs.gov/nawqa/sparrow/ | Adapted in Aotearoa by NIWA (Sandy Elliott)<br>Applied nationally and regionally for catchment planning and policy development | Schwarz (2008)<br>Schwarz et al (2006)<br>Elliott et al (2008) | Free as part of CLUES (Elliott et al, 2016)<br>US version available from USGS | Semi-distributed sub-catchment-scale model Implemented in Aotearoa as part of the CLUES model framework (Elliott, Alexander et al, 2016)<br>SPARROW and CLUES are actively maintained |
| SSYE (Suspended sediment yield estimator) and NZSYE* (New Zealand Sediment Yield Estimator) | NIWA | https://niwa.co.nz/freshwater/management-tools/sediment-tools/suspended-sediment-yield-estimator | One-off national model application for the Ministry for the Environment in 2011<br>NZSYE update applied nationally in 2019 | Hicks et al (2011)<br>Hicks et al (2019b) | Not publicly available<br>Outputs from SSYE and NZSYE are available from Ministry for the Environment | SSYE was superseded by NZSYE |
| WANSY* | NIWA | – | Application and recalibration of the SSYE model to the upper North Island | Hoyle et al (2015)<br>Haddadchi and Hicks (2016) | Not publicly available | WANSY was integrated into a bespoke version of CLUES for the Northland, Auckland and Waikato regional authorities |

Source: Adapted and updated from Elliott et al (2014). More details at Interoperable Models WIKI.
* Not included by Elliott et al. (2014)

Table 5-3: **Dynamic models applied in Aotearoa to estimate sediment load generation and transport**

| Model | Developer | URL | Aotearoa context | Key references | Status and availability | Comments |
|---|---|---|---|---|---|---|
| GLEAMS Groundwater Loading Effects of Agricultural Management Systems Model | USGS | – | Has been applied in Aotearoa for catchment planning (eg, to assess the effects of earthworks on sediment loading during urban development) | Knisel and Davis (2000) | No longer maintained or available for download | Sediment loss modelled using the Universal Soil Loss Equation (USLE) |
| LSPC* (Loading Simulation Program C++) | United States Environmental Protection Agency (USEPA) | https://cfpub.epa.gov/si/si_public_record_Report.cfm?Lab=NERL&dirEntryId=75860&CFID=22884508&CFTOKEN=98267566 | Underlying component of the Auckland Council Freshwater Management Tool | Tetra Tech Inc. (2009) | Free to download<br>No updates since 2009, but still in use as part of the BASINS model framework | Process-based erosion simulation where daily surface erosion is a function of soil type and rainfall. Bank erosion is modelled using a power function of flow rate |
| SHETRAN | School of Civil Engineering & Geosciences, Newcastle University | https://research.ncl.ac.uk/shetran/ | Applied by NIWA and Landcare Research to test catchments | Wicks and Bathurst (1996)<br>Ekanayake et al (2006)<br>Elliott et al (2012) | Free to download in two formats<br>Well documented | Grid-based model (variable resolution from 50 to 200m)<br>Primarily for smaller catchments from source to sea |
| SWAT (Soil and Water Assessment Tool) | United States Department of Agriculture (USDA)<br>Maintained by Texas A&M University | https://swat.tamu.edu/ | Applied by NIWA for sediment in the Toenepi catchment, Waikato | Neitsch et al (2009)<br>Gassman et al (2010)<br>Hoang (2019) | Open source, free download for several platforms. Well maintained and documented. New version (SWAT+) in development. World-wide user base | Used for both water quality and quantity applications<br>Sediment loss modelled using the Universal Soil Loss Equation (USLE) |

| Model | Developer | URL | Aotearoa context | Key references | Status and availability | Comments |
|---|---|---|---|---|---|---|
| Source | eWater | https://ewater.org.au/products/ewater-source/ | Has been applied to support catchment planning in Bay of Plenty and Wellington eg, Blyth (2018) | | Proprietary | Standard method is daily disaggregation of SedNet mean annual loads (ie, dynamic SedNet or D-SedNet). This Australian model has not been customised for NZ landscapes, unlike SedNetNZ |
| WEPP (Watershed Erosion Protection Project) | USDA | https://data.nal.usda.gov/dataset/water-erosion-prediction-project-wepp | Adapted in NZ by University of Canterbury (Tom Cochrane) https://teamwork.niwa.co.nz/display/IFM/WEPP | Ascough Ii et al (1997) | Free to download<br>Well maintained and documented | Specifically developed as a sediment modelling tool<br>Process based erosion simulation |

Adapted and updated from Elliott et al (2014). More details at Interoperable Models WIKI.

*Not included by Elliott et al (2014).

### 5.1.3 Representation of ESC measures in models

For the NPS-FM sediment workflow, both steady-state and dynamic models can be used to map sediment sources (and hence focus erosion mitigation efforts) and at least potentially to simulate the impact of ESC measures (section 7).

The simplest method is to apply a load reduction factor (LRF) that represents the percentage of sediment removal associated with an ESC measure.

This method has been applied using GLEAMS, SedNetNZ, NZEEM, CLUES and NZSYE (Semadeni-Davies, 2012; Semadeni-Davies and Elliott, 2012; Basher et al, 2016; Hicks et al, 2019b; Semadeni-Davies et al, 2020). LRFs are also used for some mitigations in dynamic models (eg, Waidler et al, 2011b).

Process-based methods, such as simulation of sediment settling in detention ponds (Persson et al, 1999; Persson and Wittgren 2003), or decay curves based on detention times, can be used in dynamic models.

The challenge for modelling ESC devices is in determining their performance and, where required, obtaining suitable data for the model calibration and testing. Information on the performance of ESCs used in Aotearoa is in section 7.

Measures that result in land use change (eg, afforestation and reversion of pasture into scrub) can be modelled by changing input data. However, the method used will depend on how the model represents land use (eg, in HRUs, lumped by node or sub-catchment or as a grid-cell value).

### 5.1.4 Model application examples

The following case-study applications of the SedNetNZ and NZSYE (steady state) models and the SWAT (dynamic) models are presented as examples, not recommendations. They demonstrate different modelling approaches, their data requirements, level of complexity, and the information they return.

SedNetNZ and NZSYE are gridded steady-state models. Although they have the same model purpose and use much the same input data (eg, land cover,[18] mean annual rainfall, erosion terrain[19] and slope), their complexity, resolution, data requirements and outputs differ.

In contrast, SWAT is a dynamic model which requires greater set-up and run times and has more data requirements.

#### SedNetNZ

SedNetNZ (De Rose and Basher 2011; Dymond et al, 2016) mixes physically and empirically based models to simulate a range of hillslope and channel erosion processes that contribute sediment to each stream link in a river network. It was adapted for Aotearoa from the Australian SedNet model (Prosser et al, 2001; Wilkinson et al, 2006; Wilkinson et al, 2009).

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[18] Land cover for both models has been derived from the Land Cover Database (LCDB), however the LCDB version used varies between models.

[19] Erosion terrain classes were developed by Manaaki Whenua/Landcare. The classes represent different combinations of slope, rock-type, soils, and dominant erosion processes.

Although it is a steady-state model, SedNetNZ's mean annual load outputs have been disaggregated using a rating-curve approach to provide daily sediment loads for use in eWater Source (Blyth 2018).[20] This is the same approach that Wilkinson et al (2014) used to disaggregate sediment loads estimated by the Australian SedNet model.

More recently, an alternative approach to daily disaggregation has been applied with SedNetNZ in the Bay of Plenty region (Vale et al, 2021). This apportions sediment loads generated by landslide erosion to triggering storm events. It accounts for the post-event recovery phase during which sediment loads tend to remain elevated for several years (Vale et al, 2021).

The episodic nature of storm events that initiate landslides means that their contribution to temporal sediment loads cannot simply be apportioned across the entire flow record when disaggregating mean annual loads.

SedNetNZ estimates mean annual sediment loads for each of five erosion types, in separate model modules:

- **Surficial erosion:** soil erosion due to sheet-wash, rill and inter-rill processes. This is simulated using an Aotearoa modification of the universal soil loss equation (NZUSLE; Dymond et al, 2010) as a function of mean annual rainfall, slope, soil erodibility and land cover (represented by four classes, wooded, herbaceous/pasture, bare earth and other, eg, urban).
- **Landslides:** the most common form of erosion in Aotearoa hill country, landslides are typically shallow failures of about 1 m depth and moderate areal extent (median scar size 50–100 $m^2$) based on recent satellite and aerial imagery analysis of rainfall-initiated shallow landslide erosion (Smith et al, 2021).
- **Earthflows:** the slow movement of soil and associated regolith (a region of loose unconsolidated rock and dust that sits atop a layer of bedrock) along marginal shear planes such as for erosion terrains underlain by crushed mudstone and argillite lithology.

  This is simulated for earthflow erosion terrains as an empirical function of the terrain characteristics, soil characteristics and the mean length of streams touching earthflow toes per unit area.
- **Gully erosion:** erosion initiated at gully heads as a function of water flow. This is simulated for gully erosion terrains as a function of the cross-sectional area and length of the gully, and the time since gully initialisation.
- B**ank erosion:** This is calculated by stream reach as a function of the length, estimated mean annual flow, bank migration rate and bank height for each reach.

  The bank erosion component of SedNetNZ has been updated and now includes spatial representation of additional factors that influence bank erosion, such as erodibility, channel sinuosity, and riparian woody vegetation (Smith et al, 2019).

  The bank erosion load estimate is reduced five-fold where there is riparian retirement. This is an empirical factor assumed constant for all planting types.

[20] This approach effectively uses mean annual load estimates from SedNet to calibrate a dynamic sediment load model that is underpinned by a hydrological model. The hydrological model produces spatially distributed flow records, which are converted to sediment loads through rating curve functions relating sediment load and flow.

The model outputs for land erosion are raster layers of the estimated long-term average generation of sediment from each grid-cell. These are accumulated into each stream link and then downstream, while accounting for overbank sediment storage on floodplains and sediment trapping in lakes.

By modelling different types of erosion separately, SedNetNZ can indicate which processes are most important for sediment generation in a specific area, and therefore should be targeted. However, SedNetNZ also has greater data and model set-up requirements than NZSYE.

## SedNetNZ application and examples

SedNetNZ has not been applied nationally to date, but has been used in Hawkes Bay, Waikato, Kaipara/Northland, Manawatū–Whanganui, Taranaki, Bay of Plenty, Otago and Southland (Dymond et al, 2014; Mueller and Dymond 2015; Basher et al, 2020b; Smith et al, 2020; Neverman et al, 2021; Vale et al, 2021).

A modified version of the SedNetNZ bank erosion model, however, has been applied nationally (Smith and Betts, 2021) to determine the susceptibility of stream banks to erosion. The outputs of this application can be downloaded from the Ministry's data server.

## NZSYE

The NZSYE estimator (Hicks et al, 2019b)[21] was developed for the Ministry to make national estimates of sediment loads delivered to the coast. It is an example of a purely empirical steady-state sediment load model.

The model has two components:

- A grid-based model with 1-hectare grid-cells that estimate the sediment load statistically for each cell as a function of average slope, mean annual rainfall, land cover and erosion terrain in that grid-cell.

  The combination of erosion terrain and slope can be considered a proxy for the type of erosion most prevalent from a grid-cell. Bank erosion is implicit in the calibration but is not explicitly modelled and cannot be separated from other erosion sources in the model.
- The second component aggregates the loads from cells by sub-catchment and then routes these down the River Environment Classification (REC) stream network, including through lakes and reservoirs. The model outputs are a raster layer of the mean annual stream sediment load derived from each grid-cell and in-stream sediment load by river segment.

NZSYE was calibrated nationally against river mean annual suspended sediment loads from 273 monitoring sites. These were generally estimated using measured suspended sediment concentrations and flow data using a rating curve method. Overall, the sites are largely representative of inland reaches on medium to large rivers.

## NZSYE application and examples

A semi-empirical add-on used the NZSYE model's estimated loads to coastal hydrosystems (ie, estuaries and harbours) to estimate the net sediment deposition rates in these systems.

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[21] Download model outputs by river segment from the Ministry's data server.

Since it was developed, it has been used nationally to model the potential effects of *stock exclusion* on sediment loads to inform policy development (Semadeni-Davies et al, 2020). It has also been used to support regional catchment planning in Northland (Semadeni-Davies et al, 2021).

The NZSYE outputs are available nationally via the Ministry, but do not separate out erosion types. This makes it less useful than SedNetNZ for assessing mitigation strategies that target specific erosion types. However, model overlays can be used for this purpose.

For example, in Semadeni-Davies et al (2020, 2021), the effects of stock exclusion by fencing and riparian planting, which largely targets bank erosion (Hughes, 2016), were simulated using a statistical relationship between the setback width of the riparian margin and total sediment loads delivered to streams from all sources (Sweeney and Newbold, 2014).

That is, the total load from NZSYE was reduced according to the relationship to simulate the effects riparian planting. The outputs of the simulation were revised sediment loads for each REC reach and total sediment loads delivered to coast.

## SWAT

SWAT is a semi-distributed catchment model that has been applied worldwide across a broad range of catchment scales and conditions for both hydrological and water quality modelling (Gassman et al, 2007; Neitsch et al, 2009; Gassman et al, 2010). Features include:

- SWAT represents different combinations of land cover, soil and slope using HRUs. Each sub-catchment is characterised by the proportion of each HRU it contains.
- The drainage network can be supplied to the model or created by the model at a user-specified resolution from a digital terrain model.
- The model operates with a daily time-step and produces a range of outputs including time-series and summary statistics.
- The GIS platform means that model outputs can be mapped. The model also displays outputs in summary tables and charts.
- The generic SWAT estimates sediment losses only from surface erosion using the Modified Universal Soil Loss Equation (Williams, 1975).

## SWAT application and examples

A national scale SWAT version for Aotearoa was built by Parshotam (2018, 2020). However, the national model has not been customised to represent the diverse erosion processes that typically occur in Aotearoa catchments, including landslide, gully and earthflow processes.

A catchment-scale application of SWAT in Aotearoa is the simulation of nutrient and sediment concentrations and loads for the Toenepi catchment (15 $km^2$) in Waikato (Hoang, 2019).

There, the model apparently performed poorly for phosphorus and sediment, but reliably for nitrate. However, there was a lack of suitable data to adequately evaluate the model for sediment, owing to having to rely on monthly grab water samples taken for a different purpose and mostly during base flows conditions.

Challenges for SWAT use include:

- extensive data needs[22]
- reclassification of local data metrics to those required by the model, particularly for soil
- the lack of information on rural activities and land management practices.

Unless met through substantial model set-up effort, these challenges can present significant uncertainty in model output. However, these challenges are more or less universal for any model application.

The preceding examples illustrate the need to use multiple selection criteria when identifying the appropriate model for each application. The main criteria are listed in 5.1.1.

### 5.1.5 Model uncertainty and error

**Model uncertainty** refers to unknown model reliability stemming from choice and representation of model input and outputs; model structure and the simplification of complex physical, chemical and biological processes; and the choice and calibration of model parameters.

Different models contain different approaches to erosion processes and sediment transport, each with its own set of simplifying assumptions. The choice of processes generally included playing off the need for model accuracy against data availability, and against set-up and run times.

**Model error** is separate from uncertainty. It can refer to errors in the model code as well as in the input, calibration and validation data. This may be due to, for example, the accuracy and precision of data capture, data processing methods and storage.

Errors and uncertainties within a model propagate at each step in the modelling process. A small error in input data can snowball into a substantial error in outputs. Errors and uncertainties could also compensate for each other, making it much harder to detect and evaluate them.

For discussions on the sources of model errors and uncertainty in hydrological models, see Walker et al (2003), Beven (2006), Beven and Alcock (2012) and others. Methods for assessing the uncertainties and error in water quality models have been discussed by Tscheikner-Gratl et al (2019) and Harmel et al (2009, 2014).

Model complexity and implementation aside, the sources of uncertainty and error related to data include:

- **Scaling errors related to aggregation or disaggregation of model input data and parameters** (eg, Blöschl and Sivapalan, 1995).

  Tscheikner-Gratl et al (2019) state that this issue is especially apparent where models with different spatial scales are coupled. For example, seasonal differences in sediment generation are not captured in steady-state models.

  Similarly, spatial data can be represented variously as a single point location deemed to be representative of the surrounding area (assuming that spatial variability is low), as a

[22] See Parshotam (2020) for SWAT data needs in Aotearoa.

spatially weighted average value by sub-catchment or grid cell, or using some form of quasi-distribution (eg, the proportion of each HRU within a sub-catchment).

- **Uncertainties and errors in input data and data used for model calibration and testing.** For example, there are considerable errors in load estimation from monitored water quality and flow data, particularly where water quality data is restricted to monthly grab samples and may not represent the full range of flows.
- **Lack of data to run, calibrate or test models in formats, units or scales that are compatible with the model**. For instance, one of the challenges in using the SWAT model to the Toenepi catchment (Hoang, 2019) was in obtaining soil data and characterising it into classes required by the model.

  SWAT was developed in the United States using standard soil data that is not typical of Aotearoa soils. That study also noted that the sediment model component could not be adequately tested since there was a lack of suitable sediment and flow data to calculate instream sediment loads.

  Similarly, and generally, the requirement for flow data as well as sediment concentration to estimates sediment loads means that not all sediment concentration data collected for water quality purposes can be used for model testing.

  Moreover, water quality data should be available across the full range of flow values experienced. However, monthly monitoring, such as state of environment reporting, tends not to sample from high flow events.
- **Lack or under-representation of catchment characteristics in data for model calibration and testing**. This issue is particularly important for national or regional model applications.

  The paucity of water quality sites with concurrent flow data in some catchments and regions means that there can be location biases in calibration data.

  Also, the estimated loads may not represent the full range of upstream catchment characteristics in Aotearoa.
- **Lack of data and understanding for the development and representation of land management scenarios.** There is little information available in Aotearoa on the location and type of management options already in place, and poor understanding of their efficiency.

  Sources of information include industry and the biennial Survey of Rural Decision Makers, which relies heavily on self-reporting by enterprise owners.

  Research into the efficacy of different management options has tended to focus on the paddock or hill-slope scale, so the effect at the catchment scale is largely unknown.

  Also, it is likely that the efficacy of land management options is variable across the country, due to spatial differences in climate, geology and topography.

A key recommendation to reduce model uncertainty is **to improve the current level of sediment monitoring and to collect flow data concurrently** at sediment monitoring sites. This will allow load calculation so that future modelling can draw on these data for model calibration and testing.

# 5.2 Measurement of suspended sediment load

This section focuses on methods to monitor sediment load.

See national environmental monitoring standards: NEMS–Suspended Sediment (2020), NEMS–Water Quality (2019).

### Key considerations

1. Determine the monitoring objective.
2. Continuous in-situ turbidity sensor monitoring is preferred for the NPS-FM.
3. Site specific relationships between visual clarity, suspended sediment concentration (SSC) and flow can be used to estimate sediment load for the period of interest.
4. Turbidity sensors measure the concentration of fine silt and clay.
5. Acoustic backscatter (ABS) instruments measure the concentration of coarse-grained suspended sediment (coarse silt and sand).
6. Turbidity sensors and ABS instruments together provide information about the size of suspended sediment particles.

## 5.2.1 Continuous monitoring and sediment ratings

This subsection corresponds to the sediment load monitoring methods listed in table 5-1. Methods for determining suspended sediment loads are detailed in the NEMS-Suspended Sediment (2020).

These methods range from 'continuous' monitoring to the use of 'sediment ratings'. Continuous monitoring uses in-situ suspended sediment concentration (SSC)-surrogate sensors (typically turbidity or acoustic backscatter sensors) or automatic samplers (for SSC analysis).

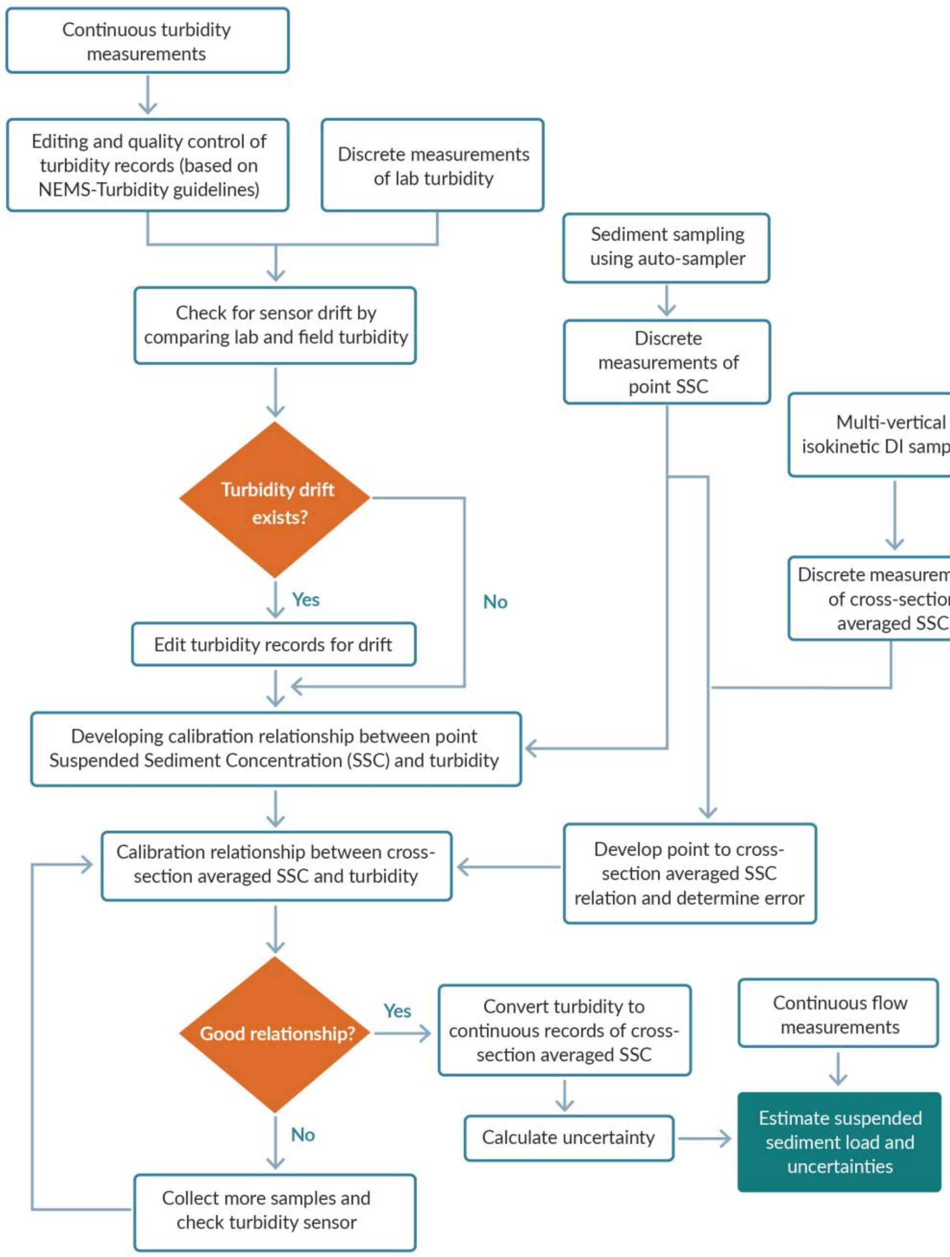
**Sediment ratings** combine discharge records with relationships between concurrently measured SSC and water discharge, or between event sediment loads and event hydrological magnitude (eg, peak discharge), to estimate time-averaged SS loads. The choice depends on the monitoring objective and resources available.

**Continuous surrogate monitoring** requires greater effort and resources, particularly since it requires some discrete sampling to relate the surrogate measure to cross-section averaged SSC, but it delivers SSC and sediment load records at high temporal resolution and reasonable accuracy.

It is suitable for estimating both mean annual loads and trends. Ongoing flow-proportional compositing auto-sampling during events also provides sub-event load resolution. But it still requires manual sampling to relate SSC at the sampling point to the cross-section average.

In contrast, sediment rating requires fewer resources because it requires only occasional automated or manual sampling. However, it is best suited for estimating long-term (multi-year) average loads since rating-curve based estimates of instantaneous SSC and event loads can involve substantial errors.

**Figure 5-2: Methodological pathway for determining suspended sediment load for a specific time period using high-frequency monitoring**




Error can occur at various steps, including:

- when collating raw turbidity records measured by the sensors
- office-based processing of raw turbidity data such as removing spikes due to electronic transients
- editing data of macrofouling and biofouling corruption, and
- managing over-ranging issues; and calibrating turbidity to cross-section average SSC.

In the NPS-FM sediment context, where resources permit, a continuously-recording in-situ turbidity sensor can service the dual objectives of providing proxy records of SSC (for both average load and trend determination) and visual clarity (for assessing the site grade against its visual clarity bottom line and visual clarity trend).[23]

Moreover, the SSC samples used to calibrate the turbidity sensor to SSC may, when also analysed for visual clarity (beam-c attenuation), be used to develop site-specific relations between visual clarity and SSC. This in turn can be used to estimate the catchment load reduction factor (equation 6, subsection 4.1.2).

The guideline for direct monitoring of suspended load is to use surrogate-SSC measurement where resources permit, following procedures detailed in the Suspended Sediment NEMS and outlined in figure 5-2.

### 5.2.2 High-frequency surrogate monitoring

Suspended sediment load estimates using discrete measurement of flow and sediment concentration have some limitations. They can have unknown accuracy due to the large temporal variability in the transport of suspended sediment during floods.

Also, the physical sampling technique may not be frequent enough to capture seasonal fluctuations in the sediment concentration and fluxes.

To overcome these limitations, surrogate technologies can be used to continuously measure suspended sediment concentration (SSC), including:

- the most commonly used surrogate technology for measuring SSC, **optical sensors** (ie, turbidity sensors, as promoted first by Gippel (1989)
- other surrogate technologies, specifically **acoustic measurements** (Topping and Wright, 2016).

The surrogate technique calibrates the collected sensor measurement (eg, turbidity) to SSC. This involves collecting in-situ water samples for SSC analysis and taking samples across a range of flows. This will require sampling during some wet weather events. Using automatic samplers can assist with this.

Figure 5-2 outlines pathways for deriving discharge-weighted SSC and suspended sediment load from high-resolution surrogate records. This includes:

- collating raw turbidity records measured by in-situ sensors
- office-based processing of raw turbidity data, such as removing spikes due to the electronic transient editing of data as a result of sensor biofouling
- detecting and adjusting over-ranging (ie, when the turbidity reading is greater than the sensor is designed to measure).

For detailed guidance on editing raw turbidity data, see NEMS-Water Quality 2019.

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[23] A caution with using a turbidity sensor for both suspended sediment load and visual clarity monitoring arises from instrument range and accuracy (which is usually a percentage of the range). This is because while the suspended load is influenced most by the high turbidity range (eg, > 1000 NTU), the median visual clarity is typically associated with the low turbidity range (eg, < 100 NTU). So, for example, a high range sensor chosen for SSC monitoring may be too inaccurate to measure the median visual clarity, or a low range sensor may completely fail to measure high SSC. Fortunately, some ‘smart’ instruments automatically rescale their range, and other instruments house both low and high-range sensors. Alternatively, you could use separate instruments for SSC and visual clarity monitoring, including acoustic back-scatter instruments for SSC.

Discrete samples collected by the autosampler over high-flow events should be analysed for both lab turbidity and SSC in the laboratory.

Check turbidity sensor drift due to instrument aging by comparing the field and laboratory turbidity data (see Hicks, 2009; NEMS-Water Quality, 2019).

After checking quality, you can establish a calibration relationship between turbidity and at-a-point SSC. As the SSC records only represent point-based SSC, if available, these should be converted to cross-section averaged SSC using isokinetic samplers across the cross-section.

The sediment concentration of these samples will be related to the point-SSC and then turbidity records to estimate continuous records of cross-section-averaged SSC and determine errors from converting point-SSC to cross-section-averaged SSC.

By multiplying continuous records of flow and derived SSC, you can determine sediment load and its associated uncertainty for the period of interest (eg, annual load, event load).

Take a similar approach to estimate sediment concentration and load from at-a-point acoustic backscatter (ABS) instruments (such as the LISST-ABS developed by Sequoia Scientific (2022). These instruments measure the concentration of coarse-grained suspended sediment, unlike turbidity meters which measure the concentration of fine silt and clay (Snazelle, 2017; Manaster et al, 2020).

These sensors can provide more accurate information where the suspended sediment is dominated by medium to coarse silt and sand. When operated together, turbidity and ABS sensors provide information about the size of suspended sediment particles.

# 6 Identifying sediment sources

This section is an overview of the methods to identify and measure, or estimate, catchment sediment sources, with a focus on sediment tracing and modelling.

Reliable quantitative information on the distribution of suspended sediment sources in catchments is required to locate major sources of sediment and develop appropriate policy responses (eg, limits on resource use and mitigations in action plans) to reduce delivery of *fine sediment* to rivers and downstream receiving environments.

See: NPS-FM 2020, clause 3.14 (setting limits on resource use) and clause 3.15 (preparing action plans).

### Key considerations

1. Divide the sediment source into upland/hillslope and instream.
2. Sediment tracing and catchment sediment modelling to map sediment sources are the two most informative methods.
3. Use field-based measurements to calibrate the two preferred methods.
4. Combine sediment load data with sediment tracing to determine the absolute sediment load from identified sources.

## 6.1 Overview of methods

In general, catchment sediment sources can be divided into upland and hillslope sources and in-stream sources. **Upland sources** often include surface erosion from hillslopes and sub-soil erosion from gullies and shallow and deep landslides. **In-stream sources** include erosion of stream banks, channel beds and bars, and floodplains. These riparian sources often involve sediment recycling after transient deposition in 'sinks' over a range of time scales (eg, as floodplain deposits are re-entrained by migrating channels or as bar deposits formed during flood recessions are re-entrained by subsequent high-flow events).

Table 6-1 sets out the tools and approaches to measure or estimate major sediment sources in catchments. These include:

- surveying and image analysis techniques
- assessing gains and losses in sediment budgets
- sediment tracing
- various spatially distributed modelling techniques (Gellis et al, 2016).

Which technique to select depends upon many factors including:

- financial resources
- timescale of measurements (from event-based to annual and multi-year measurements)
- the available timeframe for the assessment
- potential sources that need to be assessed (ie, whether the sources are in-stream processes or upland sources).

**Table 6-1: Methods to determine catchment sediment sources**

| Method | What it quantifies | Limitations and advantages | Time scale of measurements or estimates |
|---|---|---|---|
| Aerial photography and<br>Terrestrial LiDAR | Volumetric changes in bank erosion, in-stream bar deposition/erosion<br>Volumetric erosion from landslides and gullies | Only addresses pre-determined sources in specific locations within the catchment, but evaluates these at high accuracy | From single event to years |
| Bank erosion surveys | Changes in stream channel size and shape, and rate of bank erosion | Only addresses bank erosion, but evaluated to high accuracy | From single event to years |
| Sediment traps (eg, hillslope erosion traps, straw dams, see Gellis et al (2012)) | Sediment yield from upstream contributing area | Limited to assessing erosion from hillslopes where sediment traps are located. | From single event to years |
| Suspended sediment load monitoring networks (see 0 for more detail) | All potential erosion sources between adjacent monitoring sites | Lacks spatial resolution to discriminate sediment sources by erosion type and at scales finer than the network | From single event to years |
| Catchment-wide synoptic sampling of SSC during/ following run-off events | Provides a near-synoptic map-view of SSC which is assumed to be indicative of the spatial distribution of the event suspended load | Easily done, but observed SSC data are not discharge-weighted nor necessarily representative of the cross-section average SSC, and the pattern may be corrupted by hydrograph phase differences | Single events, but typically repeated over several events to assess consistency |
| Sediment tracing | Different methods can distribute load sources geographically or by erosion type. Resolves load proportion by source, and if combined with sediment load data the absolute load from sources can be quantified. | Difficult to validate estimated proportion of erosion from different sediment sources, unless validated off independent data (eg, load distribution indicated by a sampling network) | From single event to decades |
| Catchment-based steady state mean annual sediment load models<br>Examples: SedNetNZ, NZSYE | Erosion from hillslope sources and bank erosion (if physically based sub-models for bank erosion estimate are included) | Higher uncertainty than with direct monitoring and measurement techniques, but models do provide a spatially distributed sediment budget | Mean annual estimates |
| Dynamic (time-stepping) combined physically based and catchment-based sediment and water routing models<br>Examples: SWAT, LSPC | Erosion from hillslope surface sources and bank erosion (if physically based sub-models for bank erosion estimate are included), and spatially distributed SSC (in some cases for multiple size fractions) and water discharge records. SSC records can be calibrated to visual clarity. | Higher uncertainty than with monitoring techniques, but these models provide a spatially distributed continuous record of SSC and visual clarity.<br>This is the only approach that can inform on the load reduction factor in large, heterogenous catchments (section 5). Mass movement and gully erosion sources are not typically represented. | From within-event to mean annual estimates |

* Ideally using high-frequency SSC-surrogate monitoring. See 5.2.1 for more detail.

Of the methods in table 6-1, sediment tracing and catchment sediment load modelling give the most comprehensive assessment of sediment sources.

Another potential method for identifying sediment source hotspots is remotely sensed imagery. The other approaches, involving field-based measurements of erosion or instream sediment load, serve best in providing data to calibrate and validate the models or tracer results.

## 6.2 Sediment tracing

Sediment tracing (or sediment fingerprinting) techniques are widely used in catchment management to help determine the proportional contribution of catchment soil sources to fine sediment in rivers, estuaries and marine environments.

Sediment tracing techniques calculate source proportions (eg, percentage from exotic forestry, percentage from pasture) rather than absolute quantities (ie, the sediment load from each source).

By combining source proportion information with sediment load data at a monitoring site, the mass contribution of various sources to the sediment load at that site can be quantified. This combination of approaches can also resolve sediment sources of discrete events (see Vale and Dymond, 2020; Nosrati et al, 2021).

A range of sediment tracers exist, including:

- sediment properties (size, shape, colour)
- fallout radioisotopes ($^{7}Be$, $^{137}Cs$, $^{210}Pb$)
- geochemistry (eg, trace metal concentrations)
- pollen
- microbes
- magnetic susceptibility
- organic compounds.

The type of sediment tracer to select depends mainly on whether sediment sources are to be classified by landcover/land use, erosion type, or geographic location. Below are the three most widely used and reliable tracers.

- The **Compound Specific Stable Isotope (CSSI)** sediment tracing technique (Gibbs, 2008; Swales and Gibbs, 2020) – this is the appropriate approach if discriminating by land use.
- The **radionuclide tracing technique** is better to discriminate surface, sub-soil and bank erosion.
- **Geochemical tracers** are appropriate to determine contributions from different geographic sources.

The major limitation of sediment tracing is the expense and difficulty of obtaining validation data from sediment load monitoring. Overall, sediment-tracing techniques, such as geochemical and CSSI tracers, can inform catchment limit setting by:

- identifying soil-erosion hot spots in catchments by land use and sub-catchment sources
- identifying sources of in-stream or downstream (eg, estuarine) sediment deposits over different time scales, including shorter event, contemporary, and decadal timescales, that can align with or can inform regional planning
- determining river deposition footprints in downstream estuaries and coastal lakes
- providing independent data to validate outputs from numerical catchment or estuary models
- providing a means to monitor changes in source soil proportions associated with land-use management strategies over time, to assess their efficacy.

For reviews of recent developments in sediment tracing methods and their applications for sediment management, see Haddadchi et al (2013), Collins et al (2017), Laceby et al (2017) and Collins et al (2020).

## 6.3 Mapping sources with sediment load models

Steady-state and dynamic sediment load models (section 5) can be used to map and quantify sediment sources (Vale et al, 2021; Semadeni-Davies et al, 2022) and also, to various degrees, quantify the impact of ESC on downstream loads and fine sediment attributes (section 4). As table 6-1 shows, steady-state models are easier to use.

In contrast, dynamic models require substantially greater resources to build and calibrate. However, they could, after significant development and customisation for Aotearoa conditions, directly link erosion sources and treatment to in-stream sediment attributes (eg, by simulating visual clarity from SSC records).

With this capability, they could also inform on the sediment load reduction factor required to achieve the fine sediment TASs in large, heterogenous catchments.

# 7 Erosion and sediment control (ESC) practices

After mapping the dominant sediment sources (section 6) and determining the sediment load reduction required for a catchment to meet the TAS (section 4), the next step is to identify potential ESC practices and determine the extent to which they need be applied.

This section is a summary of available ESC practices commonly used in Aotearoa, based on a recent review by Phillips et al (2020). It also outlines:

- the timeframes for these measures to take full effect
- a five-step approach to implementing adaptive ESC management
- how to incorporate ESCs in catchment modelling under the NPS-FM.

See appendix C for links to sediment mitigation tools and studies.

### Key considerations

1. The performance of ESC measures is highly variable.
2. Further research is needed in Aotearoa to determine the effectiveness of set-back widths (fencing and riparian planting) on sediment, to mitigate the effects of differing land uses.
3. Land managers are encouraged to develop an adaptive ESC management programme.
4. Long-term monitoring is required to determine performance of the ESC over time, particularly as some vegetation types take a long time to reach maturity and become fully effective.
5. The simplest method for targeted mitigation in a catchment, is to apply a land reduction factor that represents the percentage of sediment removal associated with the ESC measure.
6. Effectiveness and cost of land management actions and mitigations require evaluation. Catchment groups are encouraged to keep a register of management actions and mitigations.
7. Councils and catchment groups will need to assess and weigh up investment in monitoring versus financing mitigations when faced with finite funds.

## 7.1 Available ESC practices

Table 7-1 sets out available ESC measures in Aotearoa, and their associated sediment removal efficiency. As noted in section 5, percentage sediment removal can be used as a load reduction factor in sediment load models for scenario modelling.

The ESCs in table 7-1 are grouped by erosion type. Information on the choice of ESC with respect to land use (ie, urban development, forestry, horticulture and cropping, and pastoral farming), and sources of guidance on how they should be implemented, is in Basher et al (2016) and summarised in appendix A of Phillips et al (2020).

Table 7-1 shows that restoring and maintaining vegetation are widely used for ESC, both to reduce the extent and magnitude of erosion (eg, afforestation, willow or poplar pole planting and riparian planting) and to trap sediment (eg, wetlands and infiltration or buffer strips).

Other less commonly used ESCs (not in table 7-1), include:

- improving drainage
- water control structures (eg, stream diversions, flumes, pipes and drop structures)
- debris dams
- ground recontouring
- for stream erosion, bank strengthening and sediment trapping using riprap and gabions.

The performance of ESC measures is variable due to differences in sediment properties (ie, grain size, density and shape), in implementation (eg, vegetation species used, location and extent, and the design and dimensions of any structures), in climate, and in physical characteristics of the catchment.

## 7.2 ESC time to full effectiveness

Any vegetation cover will have an effect on erosion. However, due to the time required for some vegetation types (eg, forest and woodlands) to reach maturity, it can take decades for vegetation-based ESC measures to become fully effective.

Riparian management is an often-cited means of sediment control through stream-bank armouring and stock exclusion. Riparian planting can reduce erosion rates long-term by strengthening banks and trapping fine sediments.

However, in the short term, shade from dense planting can cause the loss of undergrowth and bank-armouring vegetation, such as grasses, leading to a transient phase of increased bank erosion in small streams as the stream channel widens. The loss of undergrowth can also lead to sheetwash and rilling, which can further increase sediment loads. This process has a timeframe in the order of 20 years or more, depending on local flow conditions and storminess.

The stream widening is, essentially, a return to the natural stream morphology that preceded deforestation. Pastoral streams, which have high light environments, tend to be narrower than shaded forested streams. Several Aotearoa researchers have noted a positive relationship between stream shade and width after riparian planting (Davies-Colley, 1997; Boothroyd et al, 2004; Hughes, 2016).

A review of international literature (Anderson et al, 2019) found a similar relationship between shade and stream width for streams with small catchments. However, for streams wider than 20 metres and a catchment area in the order of 10–100 km$^2$, the relationship is reversed.

There is little information available in Aotearoa on the relationship between set-back widths and their efficacy, particularly to mitigate against the effects of plantation forestry (Basher et al, 2016). This was a key reason why Semadeni-Davies et al (2020) turned to a review of international literature by Sweeney and Newbold (2014) for a statistical relationship between set-back width and instream sediment reduction when modelling the effects of fencing and riparian planting for stock exclusion on sediment loads. Based on this review, the sediment removal efficiencies for setbacks of 1 metre, 3 metres, and 5 metres are (in percentages) 15, 34, and 46 respectively.

**Table 7-1: Common erosion sediment controls in Aotearoa and their expected long-term efficiencies by erosion process**

| Erosion process | General control principles | ESC type | Sediment removal (%) |
|---|---|---|---|
| Surface erosion | Run-off control to reduce flow rates and sediment generation<br>Sediment control to settle or trap sediment before discharge | Wetlands and sediment traps | 60–80 |
| | | Detention and retention settling ponds | 30–70* |
| | | Silt fences (urban earth works) | 99 |
| | | Riparian grass buffer strips | 40–80 |
| | | Wheel-track ripping | 90 |
| | | Wheel-track diking | 60 |
| | | Cover crops (horticulture) | 40–90 |
| | | Continuous dense, improved pasture | 50–80 |
| Mass movement (landslides and earthflows) | Control of slope hydrology and soil strength to maintain slope stability | Space-planting (full cover) | 70 |
| | | Afforestation/reversion to scrub | 90 |
| | | Debris dams | 80 |
| Gully erosion | Runoff control to reduce flow rates and sediment generation | Space-planting (full cover) | 70 |
| | | Afforestation/reversion to scrub | 90 |
| Streambank erosion | Maintain bank stability to reduce undercutting and lateral migration | Riparian fencing<br>Riparian fencing and planting | 50** |

Sources: Phillips et al (2020), Basher et al (2016)

*Adding flocculants can improve sediment (Moores and Pattinson, 2008).

**Conservative estimate. Efficiency is a function of vegetation type and the width of the riparian set-back. Sweeney and Newbold (2014) report values of ~80% for vegetated riparian buffers with a width of 10 metres.

## 7.3 Adaptive ESC management

In practice, the development of ESC programmes, often on privately owned land, is a challenging process that benefits in the long run from adapting methods through experience.

A key task is to locate the areas with the greatest need for ESC, and to tailor the measure by identifying the erosion processes at play, their drivers (eg, land use, slope, high intensity rainfall), and their spatial and temporal scales.

Schwarz et al (2020) set out a five-step plan for implementing adaptive ESC management:

1. Define and quantify the existing or potential erosion processes. Consult stakeholders affected by the erosion, and by the activity causing erosion.
2. Plan measures at different scales (eg, regional plans vs farm plans), using information from, for example, monitoring or modelling.
3. Apply the appropriate ESC measures to prevent erosion processes and maintain measures (eg, manage protective forests, restrict of forest or farming operations).
4. Apply the appropriate ESC measures to recover or mitigate erosion processes.
5. Regularly monitor and evaluate the effectiveness and efficiency of the measures, to perform maintenance or pass to a pre-engineering phase.

Despite having ample guidance for planning and implementing ESC measures, Aotearoa has significant information gaps on their performance (Phillips et al, 2020; Basher et al, 2016). This applies both to single ESC measures and those used at various spatial scales (eg, farm vs catchment) and temporal scales (single events or long-term averages).

Moreover, since vegetation-based ESC measures can take years to become established, long-term monitoring is needed to assess their performance over time.

## 7.4 Representation of ESC measures in catchment models

As outlined in section 5, both steady-state and dynamic models can be used to simulate the impact of ESC measures and estimate the cumulative downstream effect of spatially distributed control measures.

Measures involving land use change (eg, afforestation and pasture reversion to scrub) can be modelled by changing input data on land-use GIS layers. How this is done depends on how the model represents land use (eg, in hydrologic response units, lumped by node or sub-catchment, or as a grid-cell value).

The simplest method for more targeted mitigation is to apply a land reduction factor that represents the percentage of sediment removal associated with the ESC measure (table 6-1). This method has been applied using GLEAMS, SedNetNZ, NZEEM, CLUES and NZSYE (Semadeni-Davies 2012; Semadeni-Davies and Elliott 2012; Basher et al, 2016; Neverman et al, 2021; Vale et al, 2021; Semadeni-Davies et al, 2020).

However, assigning a sediment removal efficiency can be problematic. Diverse removal efficiencies have been reported, reflecting the diversity of specific locations (eg, climate, soil, slope), of mitigation designs and applications, and of the monitoring and calculation methods in different studies.

Process-based sub-models can also be used in dynamic models, such as those used for simulating sediment settling in detention ponds (Persson et al, 1999; Persson and Wittgren, 2003), or the use of decay curves based on detention times.

For example, as well as percentage removal, the SWAT model has inbuilt tools for representing both urban and rural mitigations that are **structural** (eg, grassed waterways, infiltration and filter strips, ponds and wetlands), **non-structural** (eg, inert roofing materials, street sweeping, space-planting, afforestation, mulching), and **on-channel** (eg, channel protection, riparian planting) (Waidler et al, 2011a).

The challenge for modelling ESC measures is in determining their performance and, where required, obtaining suitable data for model calibration and testing.

# 8 Monitoring and evaluation

The NPS-FM requires ongoing monitoring of suspended and deposited fine sediment attributes.

See: NPS-FM, clause 3.25; appendix 2A/table 8; appendix 2B/table 16.

## Key considerations

1. The median value from a five-year monitoring record is required to determine the 'grade' or attribute state of a site.
2. Monitoring sites should be graded annually, based on the most recent five years of data.
3. Regional councils must ensure that the monitoring regime can detect trends in attribute states (NPS-FM, clause 3.19(4)).
4. If a deteriorating trend is found to be caused by factors other than naturally occurring processes, councils must halt or reverse the degradation trend (NPS-FM, clause 3.20(1)).
5. To determine which factors are causing the trend, councils will need to maintain databases of physical characteristics, inventory of events, activities stemming from resource consents, and a register of catchment mitigation actions.
6. Ongoing monitoring of suspended sediment load from catchments undergoing ESC measures will provide a direct check on the achievement of that attribute target. It will assist with future adaptive management and provide much-needed information on the effectiveness and time-delays of those measures.

The NPS-FM requires ongoing monitoring of suspended and deposited fine sediment attributes (as visual clarity and percentage streambed cover). As outlined in section 3, this monitoring is required to:

- track progress towards achieving the TAS at identified sites within the FMU, through evaluation of current state
- identify trends in the attribute state over time, to assess if it is improving or degrading.

Councils should monitor catchment sediment loads (or estimate them using models) on an ongoing basis to evaluate the effectiveness of the regional plan's ESC resource use limits, the action plans for deposited sediment, and, where applicable, resource consent conditions.

The combined information from attribute and catchment sediment load monitoring will enable councils, with their communities and tangata whenua, to review and adapt their plans for catchment sediment management.

Attribute and catchment sediment load monitoring are discussed in detail in section 4 and section 5, respectively. This section provides guidance on monitoring in the context of site grading and assessing temporal trends in sediment attributes.

# 8.1 Monitoring and 'grading' attributes

Monitoring visual clarity and deposited fine sediment cover are addressed in section 3. The median value from a five-year monitoring record is required to determine the 'grade' or attribute state of a site.

Once the initial grade has been determined (ie, band A, B, C or D in tables 8 and 16 of the NPS-FM), monitoring sites should be graded annually based on the most recent five years of monitoring data. This enables regular tracking of progress towards the target attribute state.

# 8.2 Attribute trend assessment

Clause 3.19 of the NPS-FM requires an assessment of temporal trends in visual clarity and deposited fine sediment cover to determine whether these attributes are improving or deteriorating – irrespective of whether the site's current state grading is above or below the bottom line, or if its catchment has a current ESC plan operating.

Increasing trends in fine sediment cover and decreasing trends in visual clarity correspond to deteriorating trends, and vice versa for improving trends. If a deteriorating trend is found to be due to factors other than naturally occurring processes, then clause 3.20 of the NPS-FM requires councils to halt or reverse the degradation.

Temporal trend assessments help councils to detect freshwater degradation within the FMU, even if attribute states remain within the band limit. Councils must take action if they detect degradation, regardless of the band. Deterioration within a band may be environmentally significant and portend future deterioration of the attribute state grading.

As part of this assessment, councils should also record the values of baseline, current, and target, attribute states at a monitoring site, with a range that accounts for natural variability and sampling error,[24] and not just as an NPS-FM band.

The key obligations of councils are to:

- collect monitoring data of type and frequency that are consistent with attribute state monitoring (see section 3), and that are adequate for trend detection
- assess trends using that data
- identify the cause of any trend to establish if it is other than natural.

As detailed in section 3, both visual clarity and deposited fine sediment cover may be monitored through regular monthly sampling programmes. Visual clarity may also be monitored at much higher frequencies, using an in-situ turbidity sensor to collect a surrogate record of visual clarity.

## 8.2.1 Trend assessment procedure

This guidance focuses on assessing trends in discrete records (eg, monthly instantaneous measurements). It is based on Snelder et al (2021a), who developed a guidance document for councils for non-parametric trend assessments of freshwater environmental variables that are commonly measured in Aotearoa rivers.

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[24] As stated in clause 3.10 (4) of NPS-FM 2020 document.

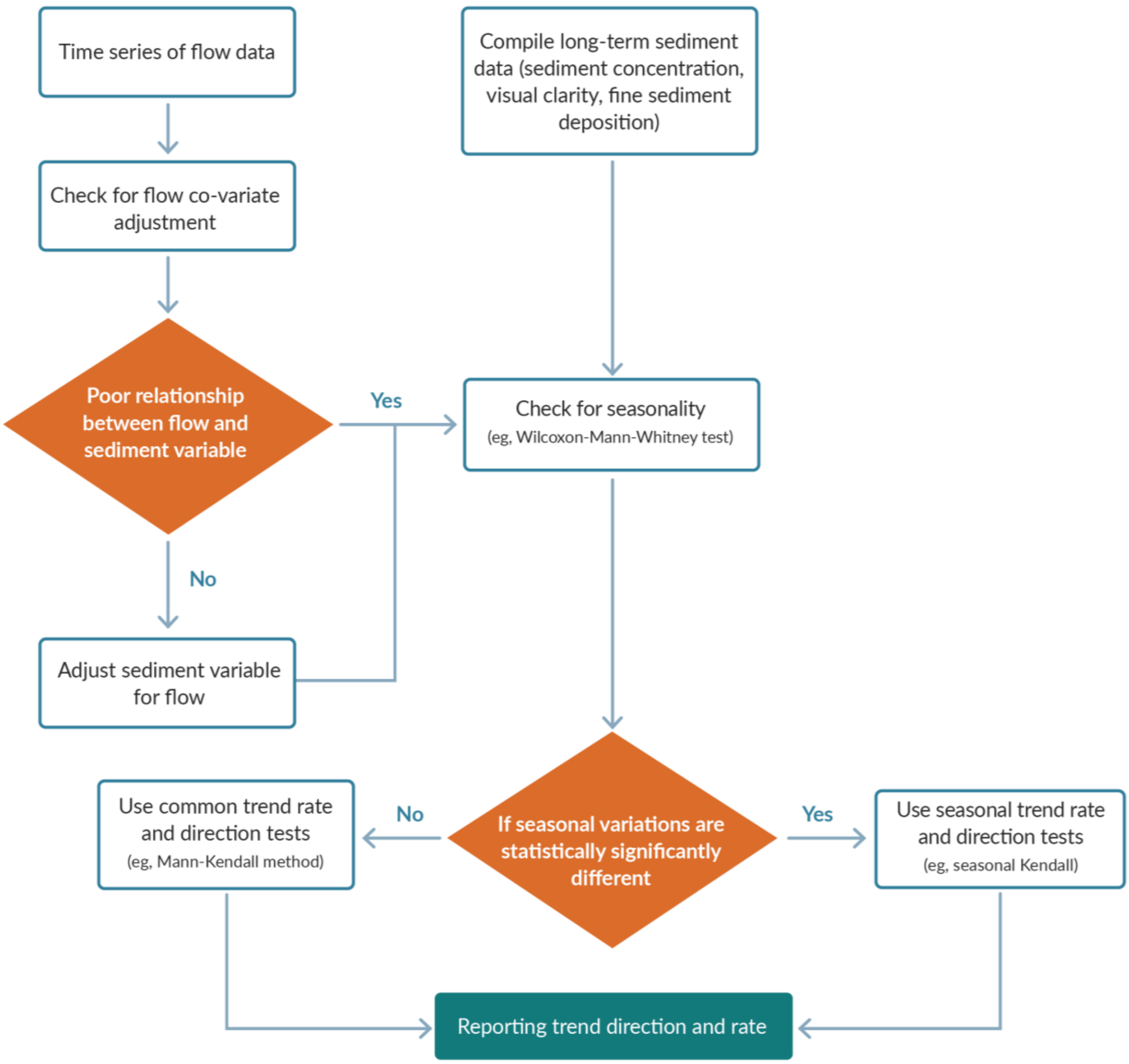
Equivalent techniques for trend analysis of high-frequency records of environmental variables, such as turbidity, are relatively new in the international literature, but indicate improved trend detection power over monthly spot sampling (Liu et al, 2020; Yang and Moyer, 2020).

This shows a further benefit of, and justification for, using high-frequency, turbidity-based surrogate monitoring of visual clarity. However, since these techniques for trend analysis from high-frequency data are relatively recent and not yet widely used, at this stage users should consult the literature for method details.[25]

The guidance here is on evaluating time series of relatively sparsely sampled, manually measured, data following the approach of Snelder et al (2021a).

Figure 8-1 shows the steps for trend analysis of sediment attributes.

**Figure 8-1: Flowchart for trend assessment analysis of fine sediment attributes**




[25] Eg, smoothing algorithms such as weighted regressions on time, discharge, and season (WRTDS) method (Hirsch et al, 2010, 2015).

**1. Compile long-term data**

The first step is to compile long-term datasets of visual clarity and deposited fine sediment cover alongside actual or estimated river flow at the time of each measurement.

River flow data are important, because fine sediment attribute measurements are commonly influenced by (or co-vary with) flow. This covariance can mask or confound the underlying trend in the attribute if it is not removed by 'flow-adjusting' the data before determining the trend.

Regression models (such as linear regression, generalised additive models, and locally estimated scatterplot smoothing) can be used for covariate adjustment and to examine the strength of the flow influence. No flow adjustment is needed if the correlation between flow and sediment attributes is poor.

**2. Check for seasonality**

The next step is to examine the data for seasonal variation. Regular seasonal fluctuations in sediment attribute data can be caused by:

- **natural** seasonal factors such as plant growth cycles and rainfall fluctuations, and
- **anthropogenic** seasonal factors such as irrigation and winter cropping.

Non-parametric tests can determine statistically significant differences among sediment data during adjacent climatic seasons. Examples of these tests are the Wilcoxon-Mann-Whitney rank-sum test (Terrio, 1996) and the Kruskal-Wallis multi-sample test (Hirsch et al, 1982).

If the attribute data shows statistically significant seasonal variation, you can use the seasonal Kendall test (Hirsch et al, 1982; Terrio, 1996) to determine monotonic time trends in fine sediment attributes.

Like the common trend tests, the seasonal Kendall test provides a single summary statistic for the entire period of record. It accounts for the effect of seasonal variation by comparing observations from the same season of different years.

If seasonal variation is not statistically significant, you can use the Mann-Kendall assessment to determine the trend direction. Its confidence level and Sen slope regression can be used to assess the trend rate. Snelder et al (2021a) detail both the Mann-Kendall and Sen slope regression methods. For other trend assessment methods see Helsel et al (2020).

**3. Report trend direction and rate**

After analysing the trend in sediment data, the final step is to report trend direction and rate, together with their confidence levels. For the calculation steps to estimate confidence in trend direction and rates as part of non-parametric trend assessments, see Snelder et al (2021a).

### 8.2.2 Attributing causes to trends

If a degrading trend is identified with statistical confidence (typically at a significance level of 0.05 or less), under the NPS-FM councils must establish if the cause is other than a naturally occurring process (ie, a process that is unaffected by human activity).

Examples in the NPS-FM of naturally occurring processes relevant for suspended sediment are:

- naturally highly coloured brown-water streams
- glacial flour-affected streams and rivers
- selected lake-fed REC classes (particularly warm climate classes) where low visual clarity may reflect autochthonous phytoplankton production.

Potential non-natural causes include human activities that directly mobilise soil and make it more susceptible to being mobilised by natural forces. These include some types of land use (eg, pastoral farming on erodible land, forest-harvesting), land use change (eg, dairy conversion, urbanisation, deforestation), earthworks (eg, for roads and subdivisions), and land drainage schemes that may, for example, accelerate run-off and channel erosion.

The natural/unnatural distinction is not always straightforward. Anthropogenic factors can compound the effects of natural weather and tectonic events to produce phases of more severe erosion (eg, when forest removal increases the risk of hill-slope erosion from cyclonic rainfall).

Unfortunately, attributing cause to trend in environmental monitoring remains in the research domain. Other than Snelder et al (2021b), little research has been done in Aotearoa and no single attribution method has been recommended or adopted. In the meantime, councils should:

- Maintain databases of catchment physical characteristics (eg, land use, land cover) and correlate any temporal changes with trends observed in the sediment attributes. This correlation should also allow for time lags between the anthropogenic catchment changes and in-stream effects.
- Keep an inventory of large storms, floods and earthquakes that have associated widespread erosion. These typically induce a multi-year transient surge in catchment erosion and sediment delivery that can confuse or compound the signature of any concurrent anthropogenic changes in catchment erodibility (eg, Basher et al, 2020b describe the effects on deposited fine sediment cover of large storms over both native and commercial exotic forests in the Motueka catchment).
- Catalogue the spatial extent of the impacts of all large erosion events, whether perceived to be natural or not. This information could assist, for example, the interpretation of observed changes in deposited fine sediment cover that may result from the migration of a sand slug downstream after a large erosion event.

## 8.3 Suspended load monitoring

Although not explicitly required in the NPS-FM, monitoring the suspended load at the FMU monitoring site is useful in two ways:

- to directly check on the effectiveness of ESC in general across the catchment; and
- to compile ‘experience’ data to inform future adaptive management.

## 8.3.1 Monitoring load reduction targets

A fundamental intermediate step that underpins the pursuit of the TAS is to set a reduction target for suspended sediment load. Ongoing monitoring of the load from catchments undergoing ESC work will provide a direct check on reaching the TAS. Section 5 addresses methods to monitor suspended loads.

The most appropriate sediment load statistic to evaluate is not straightforward. At first thought, the sediment load statistic most closely related to the five-year median of the fine sediment attributes would appear to be the five-year mean load.

However, studies of long-period suspended sediment records in Aotearoa (eg, Hicks et al, 2021) show large inter-annual variability in sediment loads associated with hydrological variability such that a five-year mean carries significant sampling error. This means, for example, that it may not be possible to identify with confidence anything less than a 50 per cent reduction in load using a five-year mean load.

There are two alternatives. The first is to **extend the averaging time base** (eg, to the 10-year mean load) to reduce the uncertainty. But the longer wait for a result renders the monitoring less useful and out of step with the five-year time-base of the sediment attribute monitoring.

The second alternative is to **'neutralise' the effect of hydrological variability** by analysing for time-trends in sediment rating relationship data. There are two ways of approaching this:

- The sediment rating may be between the instantaneous SSC (C) and water discharge (Q), $C = aQ^b$. In this case, the trend analysis tracks over time the shift in the rating offset parameter *a* from a baseline value.
- Alternatively, you can use event load ratings, $L = aQ_p^b$, which relate event sediment load (L) to an index of event hydrological magnitude (typically peak discharge, $Q_p$).

In either case, allowing that the rating exponent *b* remains steady over time, the coefficient *a* is directly proportional to the time-accumulated (or averaged) load. Hicks et al (2021) give a case example of analysing event load ratings for time trend.

## 8.3.2 Refining the knowledge base on the efficacy of sediment load reduction methods

In addition to measuring progress towards achieving the TAS, a second, strategic, benefit of monitoring sediment loads from catchments undergoing ESC measures comes from improving the knowledge base on the effectiveness and time-delays in those measures.

The current state-of-knowledge on erosion control effectiveness is imperfect, at the national and catchment scale. New monitoring data collected by individual councils should not only help them refine their own future ESC planning in an adaptive management regime, but should also, when collated and analysed nationally, refine national guidance.

# 9 Managing the effects of fine sediment in estuaries

The NPS-FM requires that, when managing freshwater, land use and development in catchments, councils must also avoid, remedy, or mitigate adverse effects (including cumulative effects) on the health and wellbeing of downstream receiving environments, including estuaries.

This section focuses on measurement and modelling methods to inform the setting of limits on resource use in upstream catchments that will protect estuarine ecosystem health.

See: NPS-FM, clause 3.5, Integrated Management.

## Key considerations

1. Fine sediment, particularly silt and clay, drive the adverse biophysical effects in estuaries.
2. The recommended default guideline value (DGV) for NZ estuaries is 2 mm yr-1 above the natural annual sedimentation rate for an estuary.
3. The natural sedimentation rate is defined as the rate under native-forested catchment (ie, base state). Estimates of this in estuaries can be determined from dated sediment cores.
4. Sediment accretion, that is vertical accumulation of sediment deposited on the substrate surface per unit area and/or time (mm/year) is the preferred method for measuring the rate of fine sediment accumulation in estuaries.
5. Modelling can inform limit setting for estuaries by evaluating the limits that achieve objectives (ie, a numeric attribute state) to maintain a value (eg, shellfish beds/kaimoana).
6. Modelling fine sediment source, transport, fate and effects in estuaries differs in several important respects from modelling river systems.
7. Regardless of the model chosen, a measured understanding of the estuary hydro- and sediment dynamics is required.
8. Sediment accretion plate monitoring can be used to validate modelled event sediment deposition. Seasonal and post-event measurements are required to isolate the scale of event deposition.

Fine sediment can adversely affect estuarine ecosystems. Major effects include reduced water clarity, and associated reduction in primary production, and smothering of benthic plant and animal communities. Although estuaries naturally infill with river-borne and marine sediment, the infill rates for many estuaries have substantially increased as a result of human activities.

The NPS-FM does not require councils to set measurable targets or limits for the coastal marine area (CMA), which includes land below mean high-water springs level, and estuaries. Management of the CMA is prescribed by the New Zealand Coastal Policy Statement (NZCPS, 2010) and implemented via regional policy statements.

However, the NPS-FM does require councils, when managing freshwater and land use, to avoid, remedy or mitigate adverse effects of sediment on estuaries. In doing so, councils are given no specific direction on which sediment attribute to monitor or target in estuaries.

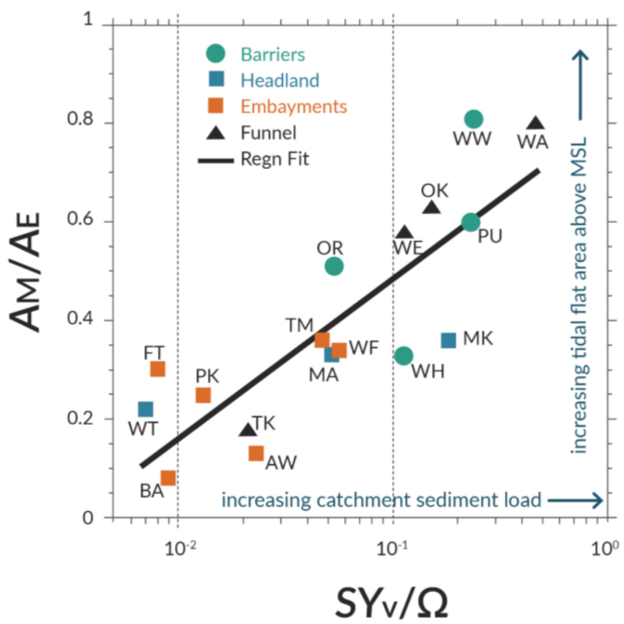
The NPS-FM sets compulsory sediment attributes for freshwater (ie, water clarity and percentage deposited sediment) but sets none for estuaries.

Although limit-setting to achieve the sediment TASs for freshwater may indirectly benefit estuaries, it may not be enough or appropriate for reaching specific objectives in estuaries. This suggests that the lack of a national objective framework for coastal receiving environments is currently a gap in the national policy framework. In the meantime, this section aims to assist councils in measuring and monitoring sedimentation in their estuaries.

## 9.1 Catchment characteristics and sediment in estuaries

Catchment characteristics play an important role in determining sediment supply to, and sedimentation rates in, estuaries. The relationship between catchment sediment supply and an estuary's sediment accommodation volume (an indication of maturity) has been demonstrated in Auckland east-coast estuaries of various geomorphic types, using the ratio of intertidal to high-tide area as a metric of estuary infilling (figure 9-1).

**Figure 9-1: Auckland east-coast estuaries: Relationship between annual catchment sediment load and estuary tidal prism volume (SYV/ Ω) ratio, and intertidal-flat above MSL to estuary area (AM/AE) ratio**




Model fit: AM/AE = 0.139*LN(SYV/ Ω) + 0.808 ($r^2 = 0.69$, $P < 0.001$). Geomorphic class (after Hume and Herdendorf 1988): (1) Barrier-enclosed estuaries with inlets formed by Holocene spits. Estuaries: Waiwera (WW), Puhoi (PU), Orewa (OR) and Whangateau (WH); (2) Headland-enclosed estuaries with inlets restricted by rocky headlands. Estuaries: Matakana (MK), Mahurangi (MA) and Waitemata (WT); (3) Coastal embayments typically with small catchments. Estuaries: Te Matuku (TM), Whitford (WF), Awaawaroa (AW), Putiki (PK), Firth of Thames (FT) and Bon Accord (BA); and (4) Funnel-shaped estuaries that have no inlet barrier, are simple or branched and form on low-energy coasts. Estuaries: Wairoa (WA), Weiti (WE) and Okura (OK). Definitions: (vertical axis) AM/AE – ratio of the tidal flat above MSL to the estuary mean high-tide surface area ($km^2$), (horizontal axis) SYV/Ω – ratio of the estimated annual average catchment sediment load ($m^3$) to the spring-tidal prism volume ($m^3$). Annual load is converted from tonnes to $m^3$ using a typical wet-bulk sediment density of 1.2 t $m^{-3}$. Source: Swales et al (2020).

The relationship is described by the relative area of intertidal flat above mean sea level (MSL) compared to the predicted annual catchment sediment load normalised by tidal prism volume based on average spring-tide range (Swales et al, 2020).

Using a bucket analogy, how rapidly the bucket fills with sediment depends on the size of the bucket and the rate of sediment addition to it.

### 9.1.1 Base state sedimentation rates

Evidence from sediment core studies indicates that sediment yields from undisturbed catchments with indigenous plant communities were an order of magnitude or so lower than the present day. Aotearoa New Zealand's estuaries very slowly infilled with sediment over the several thousand years prior to Polynesian settlement (ie, pre-1300 AD or thereabouts, Wilmshurst et al (2008)).

Sediment accumulation rates (SAR) over several thousand years during this pre-human period were in the range 0.02–1.3 mm $yr^{-1}$ (Hume and McGlone, 1986; Sheffieldt et al, 1995; Swales et al, 1997; Swales et al, 2002; Bentley et al, 2014; Handley et al, 2017). This variation in base state SAR again reflects differences in the characteristics of the catchment-estuary systems.

Although not an exhaustive data set, these base state SARs are based on 76 radiocarbon dates from cores collected in a range of estuary types. A notable gap is the data for tidal river-mouth estuaries (TRME), which account for 22 per cent of Aotearoa New Zealand's 450 coastal hydrosystems (Hume et al, 2016). Examples of TRME from the North Island's west coast include the Manawatū, Whanganui, Urenui, Tongaparutu and Mōkau rivers.

### 9.1.2 Catchment deforestation and estuary sedimentation in the recent era

Aotearoa New Zealand's estuaries have been transformed by human activities over the last several centuries, and particularly over the last ~170 years. This most recent period coincides with an order of magnitude increase in sediment loads delivered to estuaries, from catchment deforestation, mining, pastoral agriculture and urbanisation. Land-use intensification in recent decades has also been a major driver of environmental change.

The rapid increase in SAR during the historical era has seen many estuaries transformed from sand- to mud-dominated systems. This has been accompanied by the loss or degradation of ecosystems sensitive to increased water turbidity, reduced light levels and sedimentation (eg, seagrass meadows, filter-feeding shellfish) (Thrush et al, 2004).

Lead-210 ($^{210}Pb$) dating of sediment cores has been used to calculate estuary-average SAR for the post-deforestation era, to compare with base state SAR for Aotearoa New Zealand's estuaries (eg, Swales et al, 2002; Bentley et al, 2014; Handley et al, 2017). $^{210}Pb$ dating is a particularly useful tool because its time scale (up to 150 years) closely matches Aotearoa New Zealand's post-deforestation era.

$^{210}Pb$ SAR can also be validated using caesium-137 ($^{137}Cs$) for sedimentation since the early 1960s caesium-137 ($^{137}Cs$) deposition peak that is associated with atmospheric nuclear weapon tests (Ritchie and McHenry 1990). Estuary-average $^{210}Pb$ SAR reported here are time-weighted values that account for $^{210}Pb$ record length in individual sediment cores.

Data from studies of some 30 estuaries has been used to calculate an estuary-average $^{210}$Pb SAR for intertidal and subtidal flat habitats, primarily from North Island systems (Northland, Auckland, Waikato, Bay of Plenty, Greater Wellington and Marlborough). Data from coastal wetlands and tidal creeks are not included as local conditions can substantially enhance fine-sediment deposition (ie, up to several cm yr$^{-1}$, Swales et al, (2002)).

Estuary-average $^{210}$Pb SAR in these estuaries has averaged 3.2±1.1 mm yr$^{-1}$ (range: 1–5.2 mm yr$^{-1}$) over the last ~50 to 100 years. This overall estuary-average $^{210}$Pb SAR for the recent historical period is an order of magnitude higher than for the base state. In some estuaries, the increase in estuary-average SAR relative to based state has been substantially higher.

## 9.2 Measuring sedimentation in estuaries

Estuaries naturally infill with river-borne and marine sediment due to biophysical processes that favour sediment trapping. The stages of estuary development range along a continuum from youthful systems that have retained a substantial proportion of their original tidal volume, to mature estuaries that have largely infilled with sediment and have little remaining accommodation volume for sediment (eg, Swales et al, 2020).

In these mature infilled estuaries, new sediment accommodation volume is created by sea-level rise (SLR). 'Excess' sediment is exported to the adjoining coastal marine environment. In semi-mature estuaries, expansion of accreting intertidal flats progressively replaces subtidal habitats.

Measuring the sedimentation rate in an estuary provides a key metric for monitoring the following large-scale changes in an estuary as it infills:

- the loss of subtidal habitats
- the creation of intertidal habitats as average water depth decreases.

The fundamental hydrodynamic and sedimentological characteristics of estuaries change as a result of sedimentation that has accelerated over decades due to catchment deforestation, conversion to pastoral agriculture, and land use intensification. Associated increases in sedimentation and a shift from sand- to fine-sediment-dominated systems have substantially driven adverse changes in estuarine ecosystems (Thrush et al, 2004).

Encapsulated within the estuary sedimentation process are a number of environmental effects. Fine sediment, and in particular silt and clay (ie, mud, particles less than 62.5 microns), drive the adverse biophysical effects in estuaries.

Major effects include reduced water clarity, reduced primary production by plants, and the smothering of benthic plant and animal communities. This includes chronic effects of episodic deposition associated with river flood events (eg, Thrush et al, 2004).

The Ministry for the Environment commissioned NIWA to develop a default guideline value (DGV) for estuary sedimentation rate (Townsend and Lohrer, 2015). The draft DGV recognises that "there are insufficient analysed data examining the relationships between annual sedimentation rates and ecological condition to produce guidelines from local biological effects data".

Because event-based guidelines are not available, and in any case would likely be difficult to implement widely, knowledge of event-scale effects was adapted to develop a more practical DGV for managing annual sedimentation rates in Aotearoa estuaries (Townsend and Lohrer, 2015).

The recommended DGV for Aotearoa estuaries is 2 mm $yr^{-1}$ above the natural annual sedimentation rate for an estuary. The natural sedimentation rate is defined as the rate under native-forested catchment (ie, base state). It is included in the DGV as a baseline to account for estuaries or parts of estuaries with naturally high rates of sedimentation.

A number of well-established techniques exist for measuring the rate of fine sediment accumulation in estuaries. The method to adopt depends on the time scale of interest.

In the context of sediment management under the NPS-FM, *sediment accretion,* as opposed to *sediment accumulation rate (SAR)* is likely of most relevance.

Sediment accretion can be defined as the vertical accumulation of sediment deposited on the substrate surface per unit area and/or time (eg, mm per year). It is generally applied over time scales of months to years.

In contrast, SAR tends to refer to time-averaged rates, typically over annual-to-centennial or longer time scales where measurements integrate the long-term effects of erosion and deposition cycles (Swales et al, 2002; Bentley et al, 2014; Handley et al, 2017).

Buried plates are already used by many councils as a practical method for measuring sediment accretion. These provide a reference surface to measure sediment accretion, primarily on intertidal flats.

Plastic-mesh plates have been demonstrated to provide comparable data to impervious ceramic plates. They can be used in coastal wetlands where aerial roots, stems and trunks pose difficulties for sediment accretion measurement (Swales and Lovelock, 2020).

## 9.3 Modelling approaches for sediment limit-setting

Predictive models that evaluate the consequences of future options and scenarios are useful tools for informing objectives and limit-setting in estuaries. Limits are needed to manage the cumulative effects of diffuse-source contaminants, such as fine sediment (Green, 2013). Models also provide a means to consider cumulative effects.

Land-use activities in catchments are a major driver of fine-sediment effects in estuaries. Therefore, developing sediment limits for estuaries requires a catchment-to-estuary approach.

Modelling fine sediment source, transport, fate and effects in estuaries differs in several important respects from modelling river systems:

- Multidirectional transport of suspended sediment (drivers: freshwater, tides, waves, wind) vs unidirectional river flow.
- Estuaries are major fine-sediment sinks, whose trapping efficiency depends on sediment accommodation volume. For some rivers, the capacity to store fine sediment has been substantially reduced by flood-protection infrastructure.

The NPS-FM requires regional plans to have objectives which include identified freshwater values and the environmental outcomes that give effect to each value. Initial work undertaken for the Ministry on estuaries identified: (1) ecosystem health, (2) human health and recreation, (3) mahinga kai as national values (Cornelisen et al, 2017).

This study focused on identifying candidate attributes with the strongest potential for use in managing upstream (freshwater) pressures affecting the national-level estuary values. The variables shortlisted for further consideration as sediment attributes and state variables were:

- water clarity
- total suspended solids (TSS)/turbidity
- mud areal extent
- sedimentation rate
- mud content.

Cornelisen et al (2017) has further commentry on these variables.

Setting a numeric attribute state will require an understanding of the factors that influence each estuarine value. Some of the effects of suspended and deposited fine sediment on estuarine ecosystems have been documented, including threshold values for individual species (eg, cockles, sea grass) and on benthic health. However, knowledge of multiple stressors (ie, fine sediment and other diffuse-source contaminants) and cumulative effects on estuarine values is at an early stage.

Modelling can inform limit-setting for estuaries by evaluating limits that achieve the objectives (ie, a numeric attribute state) to maintain a value (eg, shellfish beds/kaimoana).

Initial work on limit-setting in estuaries for sediment has shown how catchment sediment load limits can be determined to achieve sedimentation targets (Green, 2013; Green and Daigneault, 2018). Specifically, the mean annual sediment accumulation rate (mm $yr^{-1}$) was adopted as the sediment attribute by which to achieve environmental and amenity benefits.

The mean annual SAR is indicative of a number of adverse effects of fine sediment on estuarine ecosystems (Townsend and Lohrer, 2015). Green (2013) developed a method, based on a catchment-to-estuary sediment budget, for determining catchment load limits to achieve a target SAR value (1 mm $yr^{-1}$) for single and multiple sources (sub-catchments) and sinks (estuarine sub-environments).

Development of the sediment budget is underpinned by predictive catchment sediment load and estuary hydrodynamic and sediment transport models. The catchment model is used to evaluate total event-scale sediment loads (ie, sum of loads for all particle size classes) from each catchment source under a range of conditions. Inherent/fixed factors include geology, soil type and slope, while variable factors include weather and land use.

The estuary model is used to determine event-scale sediment transport and deposition patterns for each source that contributes sediment to each estuarine sink, under a range of oceanographic and weather conditions.

The target SAR value for the estuary will be exactly achieved when balanced by a matching total mass of sediment deposited in the estuary (ie, sum of estuary sinks) for any set of source loads (Green, 2013). Validation of these models is based on decadal-scale SAR, measured using dated sediment cores (Green, 2008).

Development of methods to set sediment load limits based on other potential sediment attributes are in their infancy, and in some cases will be challenging to implement. For example, suspended sediment concentration (SSC, kg $m^{-3}$) has been demonstrated to have adverse effects on estuarine biota, and effects thresholds have been determined for various species, including submerged plants, shellfish and fish (eg, sea grass, cockle, snapper).

However, resuspension of legacy fine sediment in estuaries is a complicating factor. It implies that SSC of a water parcel at any location may not be solely attributed to river-borne fine sediment. Fine-sediment resuspension largely occurs on intertidal flats where water depth is shallow enough, even at high tide, for small estuarine waves to be effective (eg, Green and Coco, 2014). Likewise, visual clarity of estuarine waters is also influenced by resuspension of legacy fine sediment.

Although these sediment attributes are readily measured, advancing their application for limit-setting requires accounting for complex interactions between biophysical processes in predictive models. These include tides, wind, waves, cohesive-sediment behaviour (ie, flocculation) and biological factors that influence sediment resuspension (eg, feeding and burrowing activities, wave attenuation by submerged plants, increased cohesion of sediment by organic compounds).

A potential sediment attribute that is more tractable for limit-setting to achieve estuarine objectives is event deposition (mm per unit time). This variable is defined as sediment deposition that occurs during and in the immediate aftermath of catchment floods. Appropriate time scales to consider would be days to weeks, depending on site conditions.

The major benefit of developing this variable as a sediment attribute for application in predictive models is that it is more directly tied to ecological effects than is mean annual SAR (Townsend and Lohrer, 2015). Predictive models can simulate spatially varying event deposition, with and without sediment resuspension.

An event deposition attribute based solely on the deposition of an event-specific sediment load is most amenable to practical application, given the challenges of accounting for resuspension. Validation of modelled event deposition could be based on sediment accretion plate monitoring. This would ideally require seasonal and post-event measurements to isolate the scale of event deposition.

Developing numerical models to predict fine-sediment transport and fate in estuaries is a complex task. Estuarine models have been developed for a number of estuaries in Aotearoa, with varying levels of sophistication, model calibration and validation.

Many of these models have been developed for research as well as for regional councils to inform integrated catchment management of diffuse-source contaminants, including fine sediments. A brief description of the basis for these models and requirements is in Appendix B – Sediment transport models for estuaries.

There are various options for numerical models, but the validation step can never be neglected. Assumptions and constraints must always be clearly explained. Table 9-1 lists widely used process-based ocean and coastal numerical models.

Each model has advantages and disadvantages. These include suitability for particular aquatic environments (eg, biophysical processes represented, calibration data requirements, computer requirements, spatial and temporal resolution, and cost). For example, CROCO is generally used by the marine research community, and algorithms for its use in nearshore coastal environments are still being developed.

We consider the Delft3D and MIKE modelling systems the most appropriate for modelling estuary and coastal hydrodynamics and sediment transport, but there are always trade-offs in deciding on a model. Delft3D and MIKE have user support but CROCO, SCHISM and TELEMAC do not. The underpinning code and models are accessible for Delft3D, TELEMAC and CROCO but are not for MIKE.

Delft3D, TELEMAC and CROCO are open source and freely available, with community forums and support, but might not always be the easiest to use. For example, Delft3D's unstructured grid creation is complex whereas MIKE's grid creation tool is user-friendly, but the MIKE software is not free.

**Table 9-1: Popular coastal morphodynamic models**

| Software | Open source | Dimension | Description |
|---|---|---|---|
| Delft3D | Yes | 2D, 3D | Coastal, river and estuarine flows, sediment transports, waves, water quality, morphological developments and ecology<br>Structured and unstructured grids |
| SCHISM<br>Semi-implicit Cross-scale Hydroscience Integrated System Model | Yes | 2D, 3D | Coastal, river and estuarine flows, sediment transports, waves, water quality, morphological developments and ecology<br>Unstructured grids |
| MIKE+ (by DHI) | No | 1D, 2D, 3D | Coastal, river and estuarine flows, sediment transports, waves, water quality, morphological developments and ecology<br>Structured and unstructured grids. |
| TELEMAC-MASCARET | Yes | 1D, 2D, 3D | Numerical modelling of free surface coastal, river and estuarine hydraulic, sediment transport, waves and water quality |
| ROMS (CROCO)<br>Coastal and Regional Ocean Community model | Yes | 2D, 3D | Gradually including algorithms for sediment transport. An important objective is to resolve very fine scale coastal dynamics, and their interactions with larger scales. Atmosphere, surface waves, marine sediments, biogeochemistry and ecosystems |

The cost of developing these models for fine sediment will differ depending on the complexity of the physical system, the variables/attributes being simulated and the importance of determining the accuracy of outputs.

A major factor in the accuracy of hydrodynamic models is the availability and quality of bathymetry data to develop digital elevation models (DEM) that underpin model grids. Primary sources of bathymetry data are:

- LINZ hydrographic charts
- site-specific bathymetry surveys using single- and multi-beam echosounder surveys
- LiDAR.

LINZ's National Elevation Programme (Elevation data | Toitū Te Whenua Land Information New Zealand (linz.govt.nz)) provides open-source data for much of Aotearoa, including the coast. In many cases, LiDAR surveys coincide with low tides to collect elevation data for intertidal areas on coasts and in estuaries.

LiDAR provides high-quality and high-density elevation data for intertidal areas that are extremely difficult to measure from traditional boat-based surveys. DEMs for estuarine models are typically developed from all three data sources.

An important consideration is how to reduce all data, irrespective of its source, to a common vertical datum. The official New Zealand Vertical Datum 2016 (NZVD2016) defines relationships to enable elevations from earlier vertical datums or regional vertical datums to be transformed to NZVD2016.

Because bathymetry datasets have varying quality, vertical and spatial resolutions, GIS software is generally used to post-process all data and re-project to a standard coordinate system and vertical datum.

Model calibration requires measurements of key variables including hydrodynamics (ie, tidal water levels, currents, water salinity) and sediment variables (eg, SSC, visual clarity, particle size).

Validation of models requires data independent of the calibration data set. This may include measurements of variables withheld from the calibration exercise or other data that integrates the net effects of key processes over event to decadal time scales. Validation data can include measurements from sediment accretion plates and decadal-scale SAR determined from dated sediment cores (eg, Green, 2008).

The resources required to implement predictive models for estuaries will vary widely, depending on the factors above. For example, setting up a hydrodynamic model with basic calibration data (eg, water levels) using existing bathymetry data from charts and the local harbour board will be in the range of \$50,000–\$100,000.

Developing a full numerical sediment transport model that takes into account metrological forcing, river flows and sediment loads, ocean tides, currents, waves and multiple sediment-size fractions can cost in the order of NZ\$500,000 or more. The resources required depend on the quality of the calibration and validation data.

# Abbreviations

| | |
|---|---|
| BAU | Business-as-usual |
| C50 | Median suspended sediment concentration |
| CGE | Computable general equilibrium |
| CSSI | Compound specific stable isotope |
| DEM | Digital elevation model |
| ES | Ecosystem services |
| ESC | Erosion and sediment control |
| FMU | Freshwater management unit |
| FSC | percentage of deposited fine sediment cover in riverbed |
| GMP | Good Management Practices |
| GIS | Geographic information system |
| HRUs | Hydrological response units |
| IO | Input outputs |
| IWG | Intensive winter grazing |
| LiDAR | Light detection and ranging (remote sensing method) |
| LINZ | Land Information New Zealand |
| LRF | Land reduction factor |
| MCA | Multicriteria analysis |
| the Ministry | Ministry for the Environment |
| NBL | National bottom line |
| NEMS | National Environmental Monitoring Standards |
| NIWA | National Institute of Water & Atmospheric Research Ltd |
| NOF | National Objectives Framework |
| NPS-FM | National Policy Statement for Freshwater Management |
| NZVD2016 | New Zealand vertical datum 2016 |
| RDM | Robust decision-making |
| REC | River environment classification |
| RMA | Resource Management Act |
| SAM | Sediment assessment methods |
| SAR | Sediment accumulation rate |
| SHMAK | Stream health monitoring and assessment kit |
| SSC | Suspended sediment concentration |
| TAS | Target attribute state |
| TEV | Total economic value |
| TRME | Tidal river-mouth estuaries |
| TSS | Total suspended solids |
| TMotW | Te Mana o te Wai |
| V | Visual clarity |
| $V_{50}$ | Median visual clarity |
| $V_{t50}$ | Target median visual clarity |

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# Appendix A – NPS-FM fine sediment attribute tables

**Figure A-1: A copy of table 8 (appendix 2A) of the NPS-FM document for the suspended fine sediment – visual clarity attribute**

## Table 8 – Suspended fine sediment

| Value (and component) | Ecosystem health (Water quality) | | | |
|---|---|---|---|---|
| Freshwater body type | Rivers | | | |
| Attribute unit | Visual clarity (metres) | | | |
| **Attribute band and description** | **Numeric attribute state by suspended sediment class** | | | |
| | 1 | 2 | 3 | 4 |
| **A**<br>Minimal impact of suspended sediment on instream biota. Ecological communities are similar to those observed in natural reference conditions. | ≥1.78 | ≥0.93 | ≥2.95 | ≥1.38 |
| **B**<br>Low to moderate impact of suspended sediment on instream biota. Abundance of sensitive fish species may be reduced. | <1.78 and ≥1.55 | <0.93 and ≥0.76 | <2.95 and ≥2.57 | <1.38 and ≥1.17 |
| **C**<br>Moderate to high impact of suspended sediment on instream biota. Sensitive fish species may be lost. | <1.55 and >1.34 | <0.76 and >0.61 | <2.57 and >2.22 | <1.17 and >0.98 |
| **National bottom line** | **1.34** | **0.61** | **2.22** | **0.98** |
| **D**<br>High impact of suspended sediment on instream biota. Ecological communities are significantly altered and sensitive fish and macroinvertebrate species are lost or at high risk of being lost. | <1.34 | <0.61 | <2.22 | <0.98 |

The minimum record length for grading a site is the median of 5 years of at least monthly samples (at least 60 samples).

Councils may monitor turbidity and convert the measures to visual clarity.

See Appendix 2C Tables 23 and 26 for the definition of suspended sediment classes and their composition.

The following are examples of **naturally occurring processes** relevant for suspended sediment:

- naturally highly coloured brown-water streams
- glacial flour affected streams and rivers
- selected lake-fed REC classes (particularly warm climate classes) where low visual clarity may reflect autochthonous phytoplankton production.

**Figure A-2: A copy of table 16 (appendix 2B) of the NPS-FM for the fine sediment deposition attribute**

## Table 16 – Deposited fine sediment

| | | | | |
|---|---|---|---|---|
| **Value (and component)** | Ecosystem health (Physical habitat) | | | |
| **Freshwater body type** | Wadeable rivers | | | |
| **Attribute unit** | % fine sediment cover | | | |
| **Attribute band and description** | **Numeric attribute state by deposited sediment class** | | | |
| | 1 | 2 | 3 | 4 |
| **A**<br>Minimal impact of deposited fine sediment on instream biota. Ecological communities are similar to those observed in natural reference conditions. | ≤7 | ≤10 | ≤9 | ≤13 |
| **B**<br>Low to moderate impact of deposited fine sediment on instream biota. Abundance of sensitive macroinvertebrate species may be reduced. | >7 and ≤14 | >10 and ≤19 | >9 and ≤18 | >13 and ≤19 |
| **C**<br>Moderate to high impact of deposited fine sediment on instream biota. Sensitive macroinvertebrate species may be lost. | >14 and <21 | >19 and <29 | >18 and <27 | >19 and <27 |
| **National bottom line** | **21** | **29** | **27** | **27** |
| **D**<br>High impact of deposited fine sediment on instream biota. Ecological communities are significantly altered and sensitive fish and macroinvertebrate species are lost or at high risk of being lost. | >21 | >29 | >27 | >27 |

The indicator score is percentage cover of the streambed in a run habitat determined by the instream visual method, SAM2 as defined in p. 17-20 of Clapcott JE, Young RG, Harding JS., Matthaei CD, Quinn JM. and Death RG. 2011. *Sediment Assessment Methods: Protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values*. Cawthron Institute: Nelson, New Zealand. (*see* clause 1.8)

The minimum record length for grading a site is the median of 60 samples taken over 5 years of monthly monitoring, or longer for sites where flow conditions only permit monthly monitoring seasonally.

See Tables 24 and 26 in Appendix 2C for deposited sediment classes and their composition.

This attribute does not apply in river environment classes shown in Table 25 in Appendix 2C, or where clause 3.25 requires freshwater habitat monitoring.

# Appendix B – Sediment transport models for estuaries

Developing a numerical model to predict fine-sediment transport and fate in estuaries is a complex task. There is complexity inherent in understanding and measuring the initial state (eg, river, estuarine, or coastal ocean bed morphology) and the uncertainties of the governing hydrodynamic and atmospheric forcings (Khanarmuei et al, 2020).

When considering only hydrodynamic modelling (water flows and waves), the dynamic system can be corrected if the boundary, initial or forcing conditions deviate from reality at particular points in time. With sediment and morphodynamics this is not the case. A deviation from reality will ultimately change the predicted future of both the hydro- and sediment dynamics.

Nevertheless, with a thorough understanding of the underlying model assumptions, strengths, and weaknesses these models are powerful tools in understanding and predicting the physical environment, including fine-sediment transport and fate.

Investigating sediment dynamics requires an understanding of both measurements (real-world data requirements) and modelling approaches (eg, conceptual, statistical, machine learning and numerical).

Numerical methods gained popularity because they explained and approximated physical dynamics. An increase in computational power also contributed to their increased popularity. Numerical methods rely on both the discretisation of continuum dynamics (eg, approximating the physics that governs the movement of fluids) and parametrisations. The later can be described as approximating a physical process by a set of equations or results that has been shown to simulate reality appropriately.

Both these approximations have calibration factors enabling the user to finetune a model to their area of interest. Finetuning numerical models requires environmental measurements. First the hydrodynamics need to be calibrated and then the sediment-related parameters and dynamics. No matter how complex the numerical model, the effectiveness in predicting the hydrodynamics and sediment dynamics is crucial (Williams and Esteves, 2017).

Depending on the water body of interest (eg, river, estuary), the most complex numerical models will include the dynamic effects of the Coriolis force (effects relating to the Earth's rotation), meteorological forcings (winds, atmospheric pressure etc), ocean tides, river flows, suspended sediment load transport, and consideration of bedload sediment transport as well as ocean waves (waves can play an important role in the resuspension of fine sediments in the intertidal zones of estuaries).

Bedload transport alters the bed bathymetry and thus the relative depth and material available for transport. The sediment fractions within these complex models are also important as they will also contribute to the amount of material available for transport.

Model complexity is further increased by adding numerous sediment fractions. Both terrestrial and marine sediments will have a heterogeneous sediment particle size distribution. In situ sediment samples are required to gain insight into adequate numerical assumptions. These measurements will also help in understanding the differences in cohesive and non-cohesive sediment fraction. The latter is typically more associated with marine contributions. No matter the complexity of the implemented model, a measured understanding of the hydro- and sediment dynamics is required.

# Appendix C – Links for sediment mitigations

## Types of mitigation strategies

- As part of the National Science Challenge, a list of actions to include in farm environment plans has been developed. Some can be used to mitigate sediment from various land-use activities. Access the Actions to Include in a Farm Environment Plan tool.
- Manaaki Whenua Landcare Research, Sediment bottom lines: how do we get there and can we get there?

**Examples of sediment mitigation**

*Constructed wetland – Taupiri*

Learn more in the Ministry's Constructing wetlands in Taupiri story.

*Fencing waterways – Hawke's Bay*


Learn more in the Ministry's World Environment Day: Significant Hawke's Bay wetland ecosystem restored story.

*Riparian planting – Taranaki*

Learn more in the Ministry's One million $1 native trees planted for Taranaki farmers thanks to Jobs for Nature story.

*Riparian planting – Ruamahanga River*

Learn more in the Ministry's Ruamāhanga planting to deliver jobs and environmental benefits story.

## Research on mitigations

- NZ Landcare Trust
- Ministry for Primary Industries resources, including:
  - *Economic costs of hill country erosion and benefits of mitigation in New Zealand: Review and recommendation of approach*
  - *Major forestry study to assess performance of erosion and sediment control practices*
- NIWA
- Manaaki Whenua Landcare Research resources, including:
  - *Improving understanding of the effectiveness of space-planted trees in reducing shallow landslide erosion*
  - *A review of research on the erosion control effectiveness of naturally reverting mānuka (Leptospermum scoparium) and kānuka (Kunzea ericoides): Implications for erosion mitigation by space-planted mānuka on marginal hill country*
- Smarter Targeting of Erosion Control (STEC)
- Our Land and Water resources, including:
  - *Demonstrating efficacy of rural land management actions to improve water quality – How can we quantify what actions have been done?*
  - *Mai te rangi ki te whenua, mai te whenua ki te rangi: A kaupapa Māori literature review identifying land-based values and actions to benefit freshwater systems*
  - *Assessing the effectiveness of on-farm mitigation actions*
  - *How to Minimise Phosphorus Loss from Fertilised Pasture*
  - *Assessing Contaminants with Stream Order*
  - *Register of Land Management Actions*
- *Detainment Bund: a guideline for on-farm, pasture based, storm water run-off treatment*
- *The ability of detainment bunds to decrease sediments transported from pastoral catchments in surface runoff*

## Mapping

- Manaaki Whenua Landcare Research
  - *Getting to the root causes of soil erosion using high-res remote sensing*
  - *Shallow landslide susceptibility analysis supports better targeting of erosion control*
  - *LiDAR improves modelling of shallow landslide susceptibility for smarter targeting of erosion control*
  - *Application of a revised SedNetNZ model to the Oreti and Aparima catchments, Southland*

- *Modelling baseline suspended sediment loads and load reductions required to achieve Draft Freshwater Objectives for Southland*

- Ministry for the Environment
  - *Memorandum on implementing a national index for susceptibility to streambank erosion*
  - *Streambank Erosion Susceptibility Index*
- *SedNetNZ, SLUI and contaminant generation: Part 1: Sediment and water clarity*
- Our Land and Water
  - *- A More Accurate Picture of Where Soil Erosion Is Likely*

## Case studies

- Our Land and Water
  - *Traps Catch Sediment: A farmer-led study investigating the effectiveness of sediment traps to improve water quality has delivered some encouraging results.*
  - *Understanding Cause and Effect Relationships in Aotearoa's Water*
  - *Rural Professionals Fund 2020–21*
- Manaaki Whenua Landcare Research
  - *Farm and Environment Plants for Lifestylers and Small Farms: Sediment Mitigation*