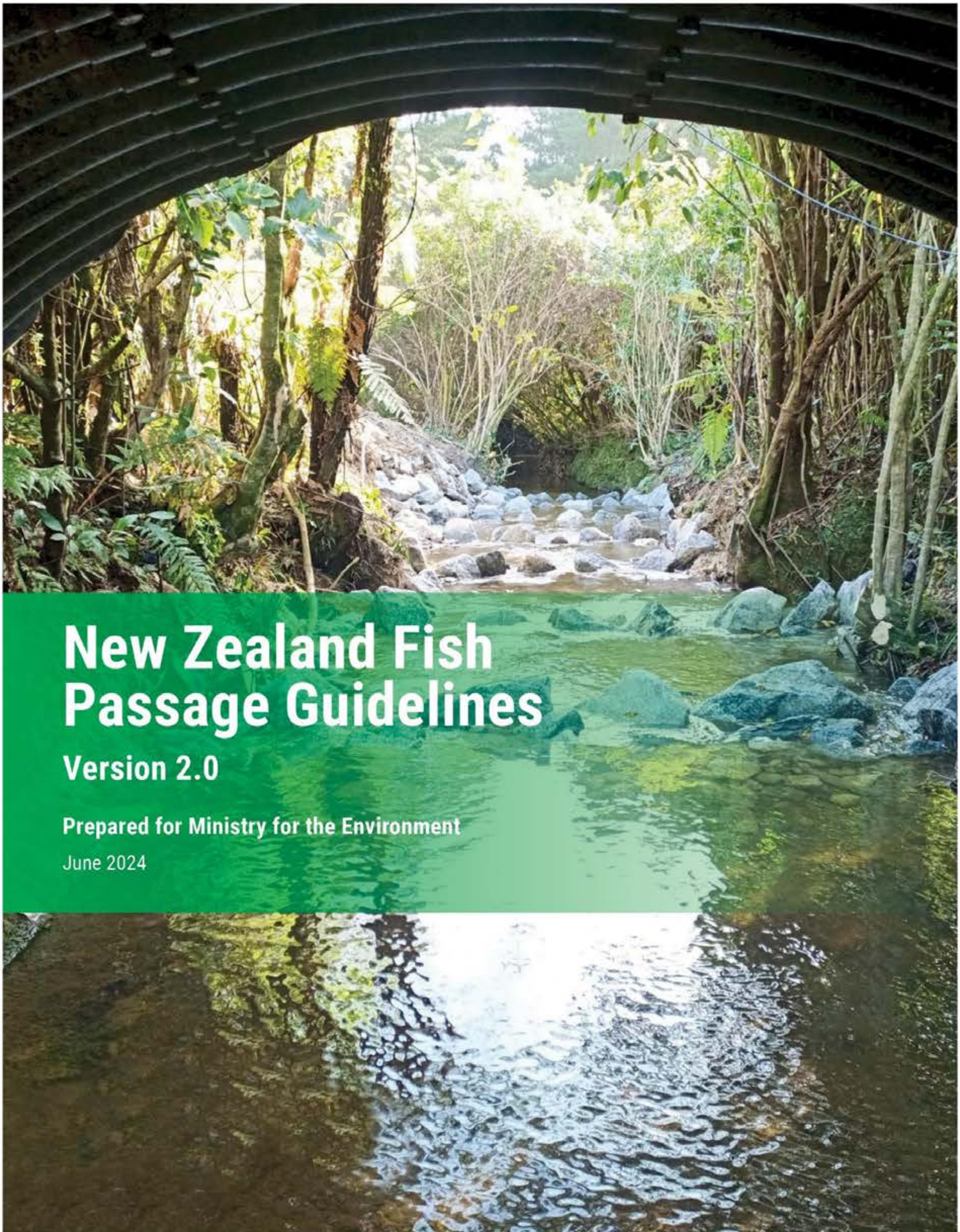

New Zealand Fish Passage Guidelines

Version 2.0

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Taihoro Nukurangi

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Manatū Mō Te Taiao

Prepared by:
Paul Franklin
Cindy Baker
Eleanor Gee
Sjaan Bowie
Michele Melchior
Eimear Egan
Liam Aghazadegan
Eugene Vodjansky

For any information regarding this report please contact:
Paul Franklin
Scientist
Freshwater Ecology
+64 7 859 1882
paul.franklin@niwa.co.nz

National Institute of Water & Atmospheric Research Ltd
PO Box 11115
Hamilton 3251

Phone +64 7 856 7026

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M J H Hickford	Reviewed by:	Mike Hickford
J Downey	Formatting checked by:	Jo Downey
M. P. Bm	Approved for release by:	Michael Bruce

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Introduction


1 Introduction

Freshwater fish are a critical component of Aotearoa New Zealand's indigenous biodiversity, with around 85% of our species found nowhere else in the world. Some of these fish species are important for sustaining cultural, recreational, and commercial fisheries and activities, and are treasured by Māori based on historical, cultural, spiritual, and ecological significance.

Most freshwater species actively or passively move between different habitats within waterways. The purpose of these movements is to access the range of habitats necessary to support different life stages (e.g., reproduction and rearing), and ecological functions (e.g., feeding or finding refuge). Some of our most widespread fish species (e.g., whitebait and eels) also undertake significant directed migrations as part of their life cycle, for example moving between the sea and fresh waters at different life stages (see Appendix A for more details). Instream infrastructure, such as culverts, weirs, and dams, can delay or prevent fish movements when adequate provision for fish movement is not provided in their design, installation, and maintenance. The consequence is a reduction in the distribution and abundance of some of our most iconic and valued freshwater species.

Fish passage refers to the ability of fish or other aquatic organisms to move unobstructed among all habitats necessary to complete their life cycles. Maintaining and restoring fish passage is essential for halting and reversing declines in freshwater biodiversity (Thieme et al. 2024). To address these issues at the scales required there is a need for greater collaboration between iwi/hapū, ecologists and engineers to enhance fish passage at instream structures (Franklin et al. 2014). These guidelines have been developed to assist infrastructure designers and managers, waterway managers, environmental officers, iwi/hapū, and local communities with understanding and promoting better management of fish passage requirements in New Zealand. The guidelines set out evidence-based approaches for providing appropriate fish passage at instream structures based on current knowledge. Due to the site-specific nature of the problem, the guidelines cannot provide a 'cookbook' of provisions for all locations. However, the general principles of good fish passage design set out in these guidelines should provide a basis for developing suitable infrastructure designs in most situations regularly encountered in New Zealand.

Te ao Māori requires an interconnected (across scales, habitats, species, and life stages) and intergenerational focus to environmental management, where taonga must be protected and enhanced for those generations not yet with us and in respect of those that have passed. **Taonga species** vary among whānau, hapū and iwi due to whakapapa connections, kaitiaki responsibilities and geographical distributions. Many taonga species have been gathered over generations and are connected to practices such as **mahinga kai**. They are also central to the intergenerational transmission of knowledge, including knowledge on the sustainable use and protection of such species and their associated ecosystems (Panelli and Tipa 2007; Panelli and Tipa 2008; Kitson et al. 2012; Nobel et al. 2016; Taura et al. 2017). **Ki uta ki tai** includes these holistic understandings of aquatic ecosystems and how the health and wellbeing of the people is intrinsically linked to that of the environment (Te Rūnanga o Ngai Tahu 2003; Kitson and Cain 2023) (Table 1-1).

The mātauranga and outcomes sought by iwi/hapū should guide all phases of fish passage design and performance, including monitoring and evaluation. When engaging iwi/hapū, note that the names of different taonga species may vary according to their life stage, depending on iwi and hapū dialect, and within different regions. For example, there is an extensive range of classifications for tuna related to appearance, coloration, season of the year, size, behaviour, locality, and palatability. It is important to acknowledge there is not “one Māori world view”. Perspectives that have been developed over time through their interactions with their environment will vary between iwi, hapū, whānau and marae.

Table 1-1: Description and definitions of some key te reo terms.

Taonga species	Mahinga kai	Ki uta ki tai
The mixed reo Māori/Pākehā phrasing ‘taonga species’ is popularised in the definitions/text penned in ‘Ko Aotearoa Tēnei’ (Waitangi Tribunal (2011) where it is described that “taonga species are the species of flora and fauna for which an iwi, hapū, or whānau says it has kaitiaki responsibilities”. Whānau, hapū, and iwi have different perspectives on what taonga species are and why (if they use this terminology at all). Many share the view that all indigenous flora and fauna are taonga species. While others advise that culturally significant flora and fauna should not be referred to as ‘taonga’ species, as all species are taonga irrespective of whether they are native or exotic (e.g. Environmental Protection Authority 2019). For some iwi/hapū, certain species are considered kaitiaki, with their presence or absence indicating the mauri of the natural environment. Some iwi/hapū have listed species of cultural significance in their Treaty settlements and other key documents (e.g. New Zealand Government 1997).	Mahinga kai, Māori customary food gathering sites and practices, is an important expression of cultural identity and values, passed down through generations (e.g. Phillips et al. 2016). Mahinga kai values include species, natural habitats, materials, and practices used for harvesting food, and places where food or resources are, or were, gathered. Mahinga kai is about the value of natural resources that sustain life, including people. It is important to manage and protect these resources in the same way that ancestors have done before. Mahinga kai areas are special places that need to be taken care of for their environmental or biodiversity significance. Some freshwater fish are mahinga kai and a staple food resource. Important examples include species such as tuna and hao (longfin and shortfin eel), kanakana or piharau (lamprey), pātiki (flounder), aua (yellow-eyed mullet), kēwai or waikōura (freshwater crayfish), paraki or porohe (smelt), mata (whitebait) and kākahi (freshwater mussel).	Ki uta ki tai can be explained as a paradigm and an ethic (Te Rūnanga o Ngai Tahu 2003). It acknowledges the connectivity of scales ‘from mountain to the sea’ as well as the reciprocal relationships between people and their ancestral environments (Ngāi Tahu ki Murihiku 2008). It directs more holistic considerations of ecosystem health and connectivity in environmental management. Ki uta ki tai is particularly important for fish passage remediation, where fish need to migrate in a variety of directions, at different stages of their life cycles. Such migrations/movements may occur locally, within and between catchments, or out into the ocean.

1.1 Purpose of the guidelines and intended audience

A key objective of these guidelines is to direct a shift away from conventional approaches to designing instream infrastructure and stream crossings towards a more holistic approach that accounts for legislative requirements to provide for fish passage. Traditional design approaches, which often focused on optimising hydraulic conveyance, run counter to the need to provide low water velocities, a diverse stream bed, and clear pathways for fish passage. Consequently, there are many structures in our waterways that do not meet legislative requirements for providing fish passage. These guidelines provide the necessary information to allow infrastructure designers to integrate the needs of fish into the design process, such that a better balance between different needs (e.g., fish passage, hydraulic conveyance, and structural integrity) can be achieved. This will help to maintain the diversity and abundance of freshwater fish and other aquatic organisms in our streams and rivers.

The intention of these guidelines is to:

- Support improved and more consistent national coordination of evidence-based fish passage management in New Zealand.
- Assist infrastructure designers, waterway managers, environmental officers, iwi/hapū, and local communities with fish passage management and how to provide appropriate passage for fish migration and movement at instream structures.
- Provide access to, and promote the adoption of, current state-of-the-art knowledge and evidence-based approaches to designing, installing, remediating, and monitoring instream structures.
- Offer practical, multipurpose, and multidisciplinary guidelines for ecologists, engineers, planners and infrastructure managers for the planning, design, implementation, remediation, and monitoring of instream infrastructure that are compatible with requirements for appropriate fish passage management and national policy.

1.2 Scope of the guidelines

This second edition of the guidelines has been expanded to incorporate a new section on providing for fish passage at dams. Consequently, the guidelines now encompass most of the commonly encountered structures in our waterways. The guidelines:

- Summarise the legislative requirements and the Government’s general policy direction that instream structures should maintain or improve fish passage, except where it is advantageous to prevent the passage of undesirable species (Section 2).
- Use an evidence-based approach to provide best-practice design criteria and guidelines on minimum design standards for installation of new structures (Section 4).
- Set out the rationale for incorporating the principles of good fish passage design into structure designs in New Zealand (Section 4.1).
- Use an evidence-based approach to recommend best-practice approaches for remediation of existing structures that impede fish passage. This includes key design requirements, common pitfalls, and approaches to ensuring retrofit solutions are fit-for-purpose (Section 5).

- Highlight the need to consider maintaining or installing barriers to manage the impacts of exotic or undesirable fish species and summarise design criteria for structures that have been successful in preventing the movements of undesirable species to protect biodiversity hotspots (Section 6).
- Set out evidence-based guidelines on providing for fish passage at dams and other high-head structures (Section 7).
- Provide recommendations for monitoring design and implementation to demonstrate the effectiveness of fish passage (Section 8).
- Include a summary of current knowledge on the passage requirements of key freshwater fish species, and an overview of structure characteristics that impede fish migrations (Appendix A & Appendix B).

The second edition of these guidelines have also been updated to include some examples of te ao Māori perspectives. This includes mātauranga related to fish passage, the strategic outcomes that Māori are seeking, and how iwi/hapū are influencing improvements for the benefit of Aotearoa New Zealand's freshwater fisheries. These additions are included as case study examples derived from a review of publicly accessible documents, including treaty settlements, plans, reports, news media, and websites.

Māori have distinct cultural knowledge, values, and perspectives that establish their identity, responsibilities, and rights to manage and use fresh water. The intention of these case studies is to encourage users of the guidelines to engage iwi/hapū to ensure their rights and interests inform all phases of fish passage planning, remediation, evaluation, and monitoring. We encourage readers to access the original documents referenced in the guidelines to ensure the information presented is understood as intended by each iwi/hapū/authorship team.

There are several examples presented in these guidelines where it is unclear if the solution put in place met the expectations of the iwi/hapū involved. There appears to be very little on-going monitoring and evaluation of these kinds of projects nationwide. We encourage users of these guidelines to talk to the iwi/hapū involved to get their feedback on whether the solutions discussed in these case studies met their longer-term expectations.

The use of macrons is the primary orthographic convention applied in these guidelines except when referencing specific examples by iwi/hapū. In such cases, double vowels have been applied or there will be no marking to lengthen vowels. The te reo Māori terms used in these guidelines are defined in Section 10.2.

The guidelines recognise the need for ongoing design development and evaluation of fish passage solutions to ensure the best outcomes for freshwater ecosystems. They acknowledge the need for innovative solutions to address connectivity barriers, but caution against the use of unproven designs that are not evidence-based and well founded in sound theory or the practical implementation of hydraulic and ecological principles. It is important to ensure that new solutions undergo appropriate monitoring and testing to validate their use prior to widespread deployment.

The guidelines do not cover all aspects of structural design and should be used in conjunction with other standard design procedures and technical documents. They also do not address:

- Non-physical barriers to migration, e.g., degraded water quality.
- The impact of artificial/heavily-modified channels on fish passage.
- Design of water intakes and diversions[1].
- The design of behavioural barriers, e.g., lights and acoustic deterrents.

In all cases, users should undertake their own site-specific design assessment and obtain specialist advice and input appropriate to the scale of the project and the value of the potentially impacted ecosystem. This should consider and recognise the limitations to our knowledge and the fact that these guidelines are based on current, best-available information that may change over time.

1.3 What has changed in the latest version of the guidelines?

The primary changes in this version of the guidelines are:

- Updates to the legislation section (Section 2) to reflect the fish passage provisions of the National Policy Statement for Freshwater Management (NPS-FM) and National Environmental Standards for Freshwater (NES-F).
- The section on fish passage objectives and performance standards (Section 3) has been revised and clarified.
- Substantial revision of the design guidelines for new culverts (Section 4.5).
- The guidelines on tide and flood gate design and remediation have been expanded (Sections 4.8 & 5.5.8 respectively).
- A new section on flood pumping stations has been added (Section 4.9).
- Updates to the guidelines on remediation of existing structures to reflect the latest evidence base (Section 5.5).
- Addition of a section on managing fish passage at dams (Section 7).
- Expansion of the section on monitoring the effectiveness of fish passage remediation (Section 8).
- Incorporation of iwi/hapū -led fish passage case studies and improved integration of te ao Māori perspectives within the guidelines.

Throughout the remainder of the guidelines, minor updates and revisions have been made to reflect the updated evidence base, and the text has been edited in places to improve conciseness and readability.

[1] For guidance on intake screening design see Hickford et al. (2023) Toward national guidance for fish screen facilities to ensure safe passage for freshwater fishes. NIWA Client report No. 2023060CH. p66

Legislative context and requirements

2 Legislative context and requirements

The Department of Conservation (DOC) and regional councils have specific responsibilities to manage fish passage in New Zealand waterways. DOC has responsibility under the Freshwater Fisheries Regulations 1983 (FFR83) while regional councils have responsibility under the Resource Management Act 1991 (RMA91), the National Policy Statement for Freshwater Management (NPS-FM) and the Resource Management (National Environmental Standards for Freshwater) Regulations 2020 (NES-F).

The fish passage provisions of the FFR83, NES-F, NPS-FM, and any regional plan rules must **ALL** be met. This means approval from both DOC **and** Regional Council could be required for the installation, maintenance, or alteration of instream structures. **Policy can be subject to change, and you should check with your regional council and DOC to make sure you have the latest information.**

Treaty settlements continue to influence the legislative foundation for the improved governance and management of Aotearoa New Zealand's freshwater ecosystems and the active implementation of rehabilitation strategies and actions to meet Māori aspirations.

2.1 The Freshwater Fisheries Regulations 1983

Under the FFR83 (Part 6, Regulations 41–50), DOC has specific fish passage responsibilities that apply to all natural rivers, streams or water, but are limited to physical barriers, i.e., dams, diversion structures, culverts and fords. These include:

- **Culverts and fords may not be built in such a way as to impede fish passage** without a permit (regulation 42(1)).
- **Culverts and fords must be maintained by the occupier**[2] **in such a way as to allow the free passage of fish and to prevent the development of fish passage barriers**, unless removed or exempted (regulation 42(2)).
- DOC **may require that any dam or diversion structure to be built has a fish facility included** and set conditions on their design and performance[3] (regulations 43 & 44).
- If a fish facility is required:
  - Every manager of a dam or diversion structure shall ensure the structure maintains adequate flow through or past the fish facility/structure, so it functions as specified at all times or periods specified within their control (regulation 45).
  - DOC may require maintenance or repair of any fish facility (regulation 46).
- That it is an offence for anyone to damage a fish facility (regulation 47).
- Approval is required for any person to make a structural change to a fish facility (regulation 48). This does not apply to any dam built pre-1984 if dispensation from the FFR was granted, or when a dam or diversion structure was built but not notified to DOC for their assessment.

The FFR83 treat a structure as either a culvert/ford or a dam/diversion structure and this is defined

[2] The term 'occupier' includes the owner of any land when there is no apparent occupier; and includes any person doing any work by contract for the occupier.

[3] Subject to the RMA91 and any determination under that Act

as being the structure that exerts the greatest control over the water (See Glossary for definition of terms). The FFR83 apply to any instream structures (e.g., floodgates, tide gates, pumping stations, water intakes) that meet the definition of a dam, diversion structure, culvert and/or ford. For example, a floodgate usually has a gate that can be opened or closed to admit or exclude water, so this gate could be a diversion structure if it diverts water, and a dam as it controls water.

No dam or diversion structure built after the FFR83 were enacted on 1 January 1984, can be retrospectively considered for approval (require a fish facility) or dispensation from the FFR83. The FFR83 generally apply to all structures built after 1 January 1984. However, regulation 42(2) (i.e., the requirement for culverts and fords to be maintained to prevent the development of fish passage barriers) applies to all culverts or fords built before and after 1984. The regulations apply to all dams or diversion structures in any natural river, stream, or water, but exclude:

- Any net, trap, or structure erected and used solely for the purpose of taking or holding fish.
- Any dam constructed on dry or swampy land or ephemeral water courses for the express purpose of watering domestic stock or providing habitat for water birds.
- Any water diversion not being incorporated into or with a dam, that is solely and reasonably required for domestic needs or for the purposes of watering domestic stock and that empties, without dead ends, into any viable fish habitat.
- Any dam or diversion structure subject to a water right issued under the provisions of the Water and Soil Conservation Act 1967 (prior to 1 January 1983) or any structure authorised by a Regional Water Board not requiring a water right that in no way impedes the passage of fish. The Water and Soil Conservation Act 1967 was the primary legislation governing the use of water resources prior to the enactment of the RMA91.

Please see Fish passage authorisations: Apply for permits (doc.govt.nz) for further information.

2.2 The Resource Management Act 1991

The Resource Management Act 1991 (RMA) is New Zealand's main piece of legislation that sets out how we should manage our environment[4]. Its purpose is to promote the sustainable management of natural and physical resources. Regional councils implement the requirements of the RMA via regional policy statements, regional plans, and the resource consenting process. Regional plans set rules governing the use of resources within a region and often contain rules relating to fish passage. The RMA is supported by several instruments that provide national direction for local decision-making. Of most relevance to fish passage are:

- National Policy Statement for Freshwater Management (NPS-FM)
- Resource Management (National Environmental Standards for Freshwater) Regulations 2020 (NES-F)

[4] Note that the RMA is currently under review by the Government.

The NPS-FM sets the overarching direction for freshwater management in New Zealand. It includes specific provisions regarding fish passage (s3.26) that regional councils must give effect to via their regional plans.

The NES-F sets out standards that regulate activities that pose risks to the health of freshwater and freshwater ecosystems. Subpart 3 addresses the impacts of instream structures on the passage of fish. Among other things, it sets out the activity status and conditions for the design and installation of new instream structures.

The NPS-FM, NES-F and regional plans have separate purposes. Regional plans must give effect to the objectives and policies of the NPS-FM. Rules in a district or regional plan may be more stringent than those in the NES-F but can only be more lenient if they relate to culverts, weirs, and passive flap gates and the rule is made for the purpose of preventing the passage of fish in order to protect particular fish species, their life stages, or their habitats. In these guidelines, an overview is provided of the NPS-FM and NES-F fish passage provisions. It is beyond the scope of these guidelines to address specific regional plan policies and rules. Consequently, please refer to the relevant regional plan or contact your regional council for local information.

The RMA also includes provisions and mechanisms for Māori involvement in decision-making regarding the Act. These include section 6(e), which requires decision makers to recognise and provide for the relationship of Māori and their culture and traditions with their ancestral lands, water, sites, waahi tapu, and other taonga. Section 7(a) introduces kaitiakitanga in relation to environmental management, which is defined in the Act as “the exercise of guardianship by the tangata whenua of an area in accordance with tikanga Māori in relation to natural and physical resources; and includes the ethic of stewardship”. The principles of the Treaty of Waitangi (Te Tiriti o Waitangi) are referenced in Section 8, which states that these principles must be considered when managing the use, development, and protection of natural and physical resources.

Mana Whakahono ā Rohe, or iwi participation arrangements (Subpart 2), provide a mechanism for iwi authorities and local authorities to discuss, agree, and record ways in which they will participate in resource management and decision-making under the Act (Ministry for the Environment 2018). Section 36B also gives power to local authorities to make joint management agreements, providing another opportunity for iwi/hapū involvement in decision-making.

2.3 National Policy Statement for Freshwater Management 2020

The current NPS-FM came into force on 3 September 2020, replacing older versions of the NPS-FM. The NPS-FM is a policy document that provides councils with direction on how to manage freshwater, including fish passage, under the RMA. Maintaining and improving fish passage is identified in the NPS-FM as a specific objective for councils (see Section 2.3.1) and they are required to develop and implement action plans to support achievement of the fish passage objective (see Section 2.3.2).

2.3.1 NPS-FM fish passage provisions

The NPS-FM Section 3.26 sets out a range of policy provisions relating to the maintenance and improvement of fish passage. It directs councils to include a fish passage objective in their regional plan(s):

> "The passage of fish is maintained, or is improved, by instream structures, except where it is desirable to prevent the passage of some fish species in order to protect desired fish species, their life stages, or their habitats."

To support achievement of this objective, councils must also include policies in their plan(s) that:

- identify desired fish species, and their relevant life stages, for which instream structures must provide passage,
- identify undesirable fish species whose passage can or should be prevented[5],
- identify rivers and receiving environments where desired fish species have been identified, and
- identify rivers and receiving environments where fish passage for undesirable fish species is to be impeded to manage their adverse effects on fish populations upstream or downstream of any barrier.

Section 3.26(4) also sets out specific matters that must be considered for consent applications relating to an instream structure. These include:

- the extent to which it provides, and will continue to provide for the foreseeable life of the structure, for the fish passage objective,
- the extent to which it does not cause a greater impediment to fish movements than occurs in adjoining river reaches and receiving environment,
- the extent to which it provides efficient and safe passage for fish, other than undesirable fish species, at all their life stages,
- the extent to which it provides the physical and hydraulic conditions necessary for the passage of fish, and
- any proposed monitoring and maintenance plan for ensuring that the structure meets the fish passage objective now and in the future.

These NPS-FM policy provisions have direct relevance for the design of new, and replacement, modification, or remediation of existing, instream structures. As far as practicable, these guidelines have been updated to reflect this policy direction.

2.3.2 NPS-FM Fish Passage Action Plans

The NPS-FM also requires that regional councils ensure that their regional plans promote the remediation of existing structures to provide fish passage where practicable. Fish Passage Action Plans (Section 3.26(6)) will set out a work programme for improving the extent to which existing structures achieve the overarching fish passage objective and set targets for remediation of existing

[5] See Section 6.2.1 for further information on fish passage and undesirable species.

structures. Importantly, the Fish Passage Action Plan **work plan must also be linked to achieving any environmental outcomes relating to the abundance and diversity of fishes**. Consequently, Fish Passage Action Plans will potentially set out fish passage objectives and performance standards (see Section 3) that will inform decision-making regarding appropriate remediation options for existing structures. Structure owners are advised to refer to local Fish Passage Action Plans as they become available. See Section 5.1 for more details on the key components of a Fish Passage Action Plan.

2.4 National Environmental Standards for Freshwater 2020

The Resource Management (National Environmental Standards for Freshwater) Regulations 2020 (NES-F) came into force on 3 September 2020. The NES-F sets out standards that regulate activities that pose risks to the health of freshwater and freshwater ecosystems.

There are three main components to the fish passage provisions:

- information requirements,
- monitoring and maintenance requirements, and
- activity status and conditions for the design and installation of new instream structures.

2.4.1 Information requirements under the NES-F

When planning works in relation to culverts, weirs, flap gates, dams or fords built after 2 September 2020, Regulations 61 to 68 of the NES-F require that certain information is provided to the regional council, whether the works require resource consent or not. The information must be provided within 20 working days after the activity is finished.

The information required by the NES-F varies depending on the type of structure but generally includes the following:

- details on the type, size, height, width, shape, and ownership of the structure,
- the location of the structure,
- details of the river such as width, depth, and velocity,
- the likelihood that the structure will impede the passage of fish, and
- information about aprons and ramps.

For full details of the information requirements for each structure type, please refer directly to the NES-F. The Fish Passage Assessment Tool can be used to capture the required information.

Fish Passage Assessment Tool

The Fish Passage Assessment Tool (FPAT) has been developed to provide an easy to use, practical tool for recording instream structures and assessing their likely impact on fish movement and river connectivity. The FPAT is available via a free mobile app or web form. Its use for providing the required information under Regulations 61 to 68 of the NES-F is endorsed by the Ministry for the Environment. Some councils have their own method for collecting the required information so please check with the relevant regional council.

2.4.2 Monitoring and maintenance requirements under the NES-F

Where a resource consent is granted for works in relation to new instream structures (culverts, weirs, flap gates, dams or fords built from 3 September 2020), Regulation 69 of the NES-F requires the resource consent to include conditions around monitoring and maintenance, including:

- monitoring and maintenance of the structure to ensure that the passage of fish does not reduce over the lifetime of the structure,
- preparation of a monitoring and maintenance plan, and
- updating the information requirements at specified intervals and when a significant natural hazard affects the structure.

These requirements for monitoring and maintenance associated with consented structures will impose additional life-time costs that should be accounted for when evaluating alternative structure designs. It is possible in some circumstances that higher up-front costs of one design relative to another may be offset by higher monitoring and maintenance costs over time.

2.4.3 Activity status and conditions for new structures under the NES-F

Regulations 70–74 set out the activity status and conditions for the placement, use, alteration, extension, or reconstruction of culverts, weirs, and passive flap gates. Specific design criteria are set out for culverts and weirs to achieve permitted activity status (see below). Where any of these conditions cannot be met, culverts and weirs are classified as discretionary activities and will require a resource consent and be subject to the Regulation 69 requirements for monitoring and maintenance. All passive flap gates are classified as non-complying activities.

Permitted activity culvert conditions

To achieve permitted activity status, a culvert must comply with the following conditions (Regulation 70(2)):

- the culvert must provide for the same passage of fish upstream and downstream as would exist without the culvert, except as required to carry out the works to place, alter, extend, or reconstruct the culvert,
- the culvert must be laid parallel to the slope of the bed of the river or connected area,
- the mean cross-sectional water velocity in the culvert must be no greater than that in all immediately adjoining river reaches,
- the culvert's width where it intersects with the bed of the river or connected area (s) and the width of the bed at that location (w), both measured in metres, must compare as follows:
  - where $w \leq 3$ m, $s \geq 1.3 \times w$:
  - where $w > 3$ m, $s \geq (1.2 \times w) + 0.6$,
- the culvert must be open-bottomed, or its invert must be placed so that at least 25% of the culvert's diameter is below the level of the bed,

- the bed substrate must be present over the full length of the culvert and stable at the flow rate at or below which the water flows for 80% of the time, and
- the culvert provides for continuity of geomorphic processes (such as the movement of sediment and debris).

Permitted activity weir conditions

To achieve permitted activity status, a weir (excluding customary weirs) must comply with the following conditions (Regulation 72(2)):

- the weir must provide for the same passage of fish upstream and downstream as would exist without the weir, except as required to carry out the works to place, alter, extend, or reconstruct the weir,
- the fall height of the weir must be no more than 0.5 m,
- the slope of the weir must be no steeper than 1:30,
- the face of the weir must have roughness elements that are mixed grade rocks of 150 to 200 mm diameter and irregularly spaced no more than 90 mm apart to create a hydraulically diverse flow structure across the weir (including any wetted margins), and
- the weir's lateral profile must be V-shaped, sloping up at the banks, and with a low-flow channel in the centre, with the lateral cross-section slope between 5° to 10°.

As far as practicable, these design conditions have been considered in developing these guidelines.

2.5 Treaty settlements and iwi/hapū environment plans

Numerous Treaty settlement arrangements contain specific provisions relating to waterways and freshwater fisheries that will inform aspects of fish passage design, performance, remediation, and monitoring in some areas. Te Haeata provides a searchable record of Treaty of Waitangi settlement commitments that may be useful for identifying and understanding relevant commitments. Case Study 1 offers some examples of specific treaty settlement provisions that may be relevant to fish passage management and should be considered by asset owners and river managers.

Some iwi/hapū have produced environmental management plans that are publicly accessible (via iwi/hapū and/or regional council websites). Generally, iwi environmental management plans (IEMP) are documents developed by iwi/hapū that identify environmental kaupapa of significance and provide details around how they expect to engage in environmental planning and decision-making processes. These plans can help practitioners better understand some of the objectives sought by iwi/hapū before they have a direct conversation with mana whenua. IEMPs can vary in style, content, spatial and temporal specificity, and can include outcomes sought, concerns, issues, objectives, methods and/or policies in relation to various environmental kaupapa including mahinga kai and fish passage.

Case Study 1: Example of treaty settlement arrangements containing specific provisions relating to waterways and freshwater fisheries.

The Waikato-Tainui Raupatu Claims (Waikato River) Settlement Act was passed in 2010 with an overarching purpose to restore and protect the health and well-being of the Waikato River for future generations. The act established an iwi–Crown co-governance structure over the Waikato River. Te Ture Whaimana (Vision and Strategy for the Waikato River) is the primary direction-setting document for the Waikato River. It sets out key objectives for protecting and restoring the Waikato River now and for future generations, including "The protection and enhancement of significant sites, fisheries, flora and fauna" (Waikato River Authority 2011). Te Ture Whaimana sits ahead of all other subordinate legislation or planning documents under the Resource Management Act (1991).

As part of the Waikato and Waipā River co-governance and co-management agreements (Waikato-Tainui (Waikato River) Fisheries Regulations 2011), Waikato-Tainui, Ngāti Tūwharetoa, Ngāti Raukawa, Te Arawa River Iwi Trust and Ngāti Maniapoto are required to develop fisheries plans. These plans set out specific objectives to protect and restore fisheries resources in the catchment and are of relevance when considering goals for restoring fish passage. For example, the Maniapoto Upper Waipā Fisheries Plan (prepared under the Nga Wai o Maniapoto (Waipā River) Act 2012) establishes objectives including:

- "The ecological functions that support the fishery of the Waipā River, are restored and protected through a holistic, integrated coordinated approach, consistent with the tikanga, kawa and mātauranga of Maniapoto."
- "Activities that result in a reduction in habitat or fish (such as habitat degradation, **fish passage**, land-based effects) are avoided, remedied or mitigated."
- "Support initiatives that will result in improved aquatic habitat that will support healthy and sustainable fisheries."

(Maniapoto Māori Trust Board 2015)

Furthermore, they seek to "advocate for fisheries habitat restoration, creation, enhancement and protection" particularly with respect to (among other things) fish passage (Maniapoto Māori Trust Board 2015).

2.6 Statutory approvals

Approval from both DOC and the relevant regional council is often required for instream structures in New Zealand's waterways, including their installation, use, alteration, or removal. The statutory requirements should be identified and considered early in the process so any constraints and associated timeframes can be factored into the project.

2.6.1 Does the instream structure require DOC authorisation?

Fish passage authorisations are required from DOC under the FFR83 for the following activities:

- existing culverts and fords that impede fish passage,
- new culverts or fords, that have been constructed to protect vulnerable species or habitat, that will impede fish passage,

- proposed damming or diversion of water that may require a fish facility[6], or
- structural modification of an existing fish facility that already has approval under the FFR.

See Fish passage authorisations: Apply for permits (doc.govt.nz).

2.6.2 Does the instream structure require a resource consent?

Instream structures often require resource consent from the relevant regional council. There are many variables that will affect whether the activity is permitted (i.e., can be undertaken without resource consent) or whether resource consent is required to approve the activity. Consequently, it is important to undertake an assessment of the likely consenting requirements early in the design process. This should consider the regulations under the NES-F and the relevant regional and city/district plans.

Regulations 70 to 74 of the NES-F operate similarly to rules in regional plans and specify the activity status of culverts, weirs, and passive flap gates under the NES-F (see Section 2.4.3).

NES-F regulations for culverts, weirs, and passive flap gates

The NES-F regulations specify the activity status for new culverts, weirs (excluding customary weirs) and passive flap gates that were not in place on 2 September 2020.

The placement, use, alteration, extension, or reconstruction of a culvert or a weir in, on, over, or under the bed of any river or connected area is permitted under the NES-F if it complies with the conditions specified under regulation 70(2) for culverts and 72(2) for weirs (see above). If the activity does not meet those conditions, then it will require resource consent as a discretionary activity under the NES-F.

The placement, use, alteration, extension, or reconstruction of a passive flap gate in, on, over, or under the bed of any river or connected area requires resource consent as a non-complying activity under the NES-F. Councils can only grant applications for non-complying activities if they are satisfied the adverse effects are minor, or granting the applications will be consistent with their regional plan’s objectives and policies.

Regional plan rules

In addition to the NES-F regulations around culverts, weirs, and passive flap gates, regional plans contain rules regarding instream structures. These rules vary across the country but generally resource consent may be required for the installation, use, alteration, or removal of instream structures, including remediation of existing instream structures for fish passage. Regional rules can be more stringent than those in the NES-F, or more lenient if they consider that impeding passage is required to protect certain fish species. Please refer to the relevant regional plan for further information.

Associated activities

Often there are other activities related to works associated with instream structures, such as earthworks or temporary stream diversions, that may also trigger the requirement for resource

[6] A fish facility is any structure or device, such as a fish pass or fish screen that is inserted in or by any waterway, to stop, allow or control the passage of fish through, around, or past any instream structure.

consent from the regional council. In some cases, these associated activities may require resource consent from the relevant city or district council.

2.7 Other statutory requirements

In addition to specific fish passage requirements, there are other statutory requirements that need to be considered in any proposals for development and management of physical structures. These can include:

- Design integrity for intended purpose and on-going management of structures and assets (e.g., Building Act 2004, Railways Act 2005, RMA91, Local Government Act 2002).
- Land status (such as landowner approval for any works on their property and on special status areas, e.g., Reserves Act 1977).
- Protection of species and habitat, for instance Section 26ZJ of the Conservation Act 1987 (CA87), which provides that it is an offence if any works (e.g., installing a structure into a waterway) disturb or damage spawning grounds of any freshwater fish.
- Fish salvage, which can often be required in construction projects within waterways. If, during any fish salvage or translocation, someone wishes to transfer and release fish into any freshwater, they are likely to require approval under Section 26ZM of the CA87 and/or regulation 59 of the FFR83.
- An archaeological authority under the Heritage New Zealand Pouhere Taonga Act 2014 for modification or destruction of any archaeological site is required for any structure/site dated pre-1900s, whether recorded or not. In addition to this, historic structures/sites of any age can be listed on the relevant district plan's heritage schedule, which will require additional consents if you wish to modify or remove the structure/site.

Fish passage objectives and performance standards

3 Fish passage objectives and performance standards

Defining clear objectives is an integral element of all instream structure design or remediation projects. The objectives define the design criteria and inform the development of the performance standards against which the effectiveness of the structure can be evaluated. Having clear and specific objectives is essential for effective design and implementing appropriate outcome monitoring.

Conroy and Peterson (2013) define objectives as “specific, quantifiable outcomes that reflect the values of decision makers and stakeholders and relate directly to the management decisions.” They also distinguish between **fundamental objectives**, that is the things that a decision maker truly values and wants to achieve, and **means objectives**, which are a means of fulfilling or achieving the fundamental objectives.

Our experience is that a priori objectives are rarely explicitly defined for fish passage projects. This contributes to ill-informed fish passage design and the absence of performance measures against which to evaluate success. In fish passage projects where a priori objectives have been identified, they are often vague and fail to distinguish between fundamental and means objectives. This leads to confusion in the design process and in measuring success.

3.1 Fish passage objectives

Objective setting should precede decisions regarding the design of new structures or remediation options for existing barriers. Effort should be taken to determine both fundamental and means objectives and understand how objectives link to, or conflict with, each other. Objectives may, among other things, reflect legal mandates (e.g., policy requirements under the NPS-FM and FFR83), community and stakeholder values, cultural needs (e.g., mahinga kai), economic values, and/or logistical considerations. We suggest that councils consider defining fish passage objectives in their fish passage action plans. Providing for fish passage will not always be the objective and in some locations, consideration should be given to non-migratory galaxiids that may be protected by man-made structures by preventing migration of undesirable species.

Fundamental objectives (green boxes in Figure 3-1) will often reflect values at a broader catchment or riverscape scale, whereas means objectives (blue boxes in Figure 3-1) will more often reflect site or structure scale performance requirements needed to achieve the fundamental objectives (O’Connor et al. 2022). In the context of the NPS-FM, fundamental objectives likely align with long-term visions (s3.3) and environmental outcomes (s3.9) that are set by councils in collaboration with tangata whenua and communities Figure 3-1). Means objectives are defined in support of achieving the fundamental objectives. The NPS-FM and FFR83 identify several means objectives with respect to fish passage that must be considered in any fish passage project (e.g., NPS-FM s3.26(1), 3.26(2d), 3.26(4b) & FFR83 R42, R43; Figure 3-1). However, further means objectives (see blue boxes in Figure 3-1) will be required to underpin achievement of both the NPS-FM means objectives and the fundamental objectives.

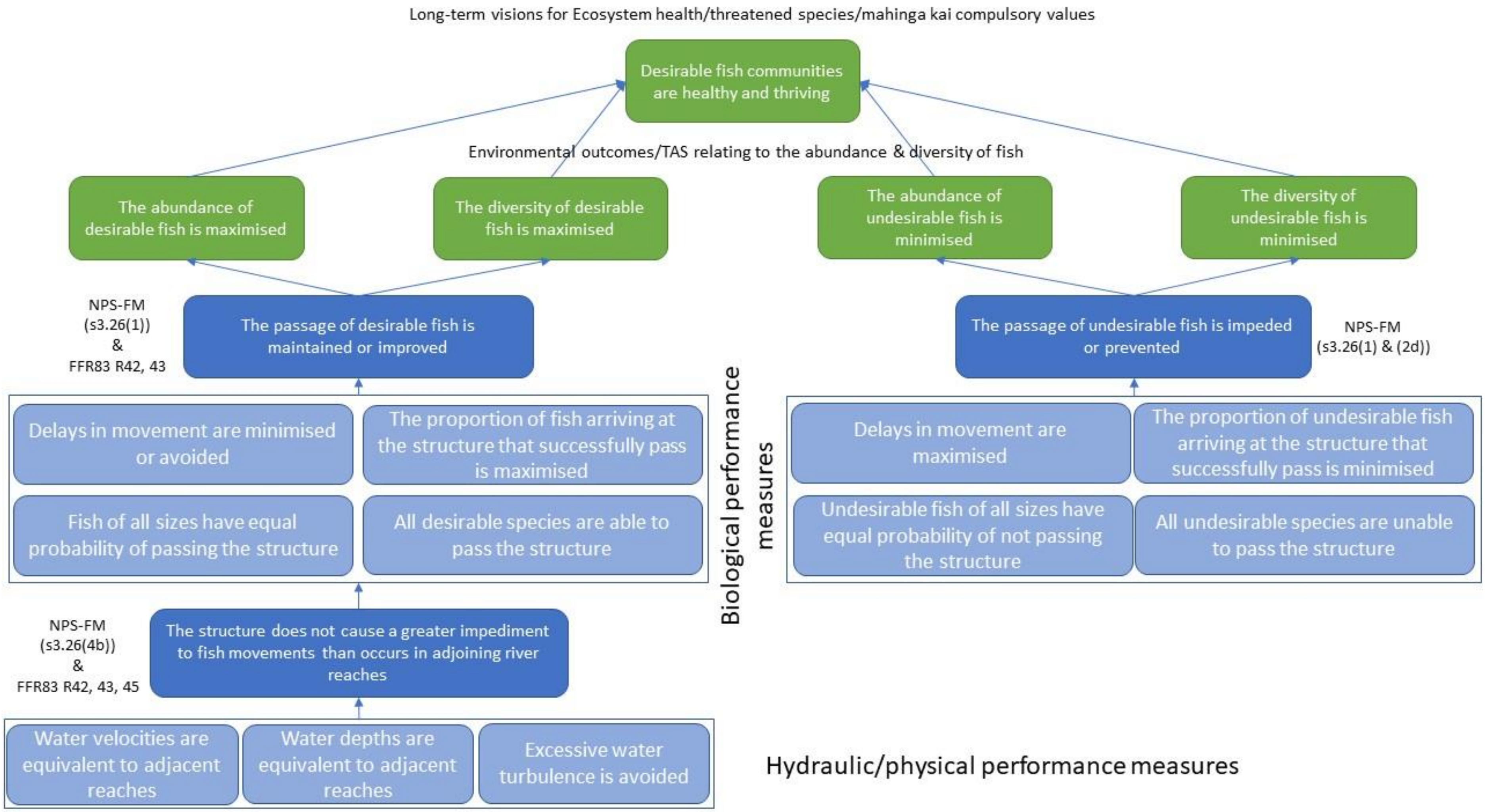


Figure 3-1: Illustration of a potential fish passage objectives network that links means objectives to achieving fundamental objectives.

The green boxes are examples of fundamental objectives, and the blue boxes are examples of possible means objectives. This objectives network is not intended to be exhaustive and will vary depending on the specific context and values for the site. References to the NPS-FM and FFR83 show links with specific provisions within those policies/regulations.

Figure 3-1 provides an example objectives network for a fish passage project. The example is focused on ecological objectives, but expanding this to consider and incorporate broader economic, social, cultural, and logistical objectives, for example, may be important for effectively evaluating trade-offs in the decision-making process (Gregory and Keeney 2002). The process of identifying objectives should involve tangata whenua and stakeholders. This will increase the legitimacy of the objectives and subsequent decision-making regarding potential solutions and performance measures.

The objectives set the boundaries for identifying alternative options for solutions that will satisfy the objectives and for defining performance measures. Status quo bias often leads decision-makers to short-cut the process of identifying and evaluating alternative solutions by defaulting to a known or easily-accessible intervention (Conroy and Peterson 2013). This can result in better alternatives being missed and/or solutions being poorly aligned with objectives and so should be avoided. It is important for decision-makers to recognise that multiple means objectives may have to be fulfilled to achieve the fundamental objectives. This is a common occurrence in fish passage management where providing passage for multiple species and life stages at the same site is a typical pre-requisite for achieving broader fundamental objectives relating to sustaining healthy upstream fish communities. Translating the means objectives into performance measures and standards (O'Connor et al. 2022) and aligning this with the development of design criteria and project monitoring and evaluation (see Section 8 for further discussion of monitoring and evaluation) is an important step for helping to evaluate alternative solutions and ensuring accountability.

Iwi/hapū-authored documents such as fisheries plans, environmental management plans and restoration plans contain a wealth of information about their desired outcomes and objectives and may be valuable in establishing fish passage objectives (Case Study 2). These plans often include lists of taonga species and/or unwanted species that may be used to identify desirable or undesirable species, respectively. For example, the Raukawa Fisheries Plan identifies tuna, kōura, piharau, kōkopu, kōaro and kāeo/kākahi as freshwater species utilised as a food source (Raukawa Charitable Trust 2012), while the Te Arawa River Iwi Trust Fisheries Plan lists tuna, koura, kākahi, kōaro/ kōkopu and morihana (goldfish) as customary taonga species (Te Arawa River Iwi Trust 2021).

Iwi/hapū used/use a variety of methods to catch different species and life stages, with techniques varying by area, season, and habitat. These activities would often be undertaken according to the maramataka (calendar that divides the Māori year into lunar months). This knowledge of past and present fisheries diversity, fishing conditions, effective methods, local habitats, and timings (e.g., migrations) can inform the setting of fish passage objectives and performance measures. How iwi/hapū choose to set fish passage objectives and assess ecosystem responses to fish passage remediation activities is likely to be rohe-, catchment- and/or species-dependent. Iwi/hapū objectives may consider the needs of all life stages (in space and time), their mahinga kai and wider strategic goals, as well as the needs of interdependent species/fish communities, such as kākahi (freshwater mussel).

Case Study 2: Examples of fish passage objectives in environmental management plans developed by iwi/hapū.

Ngāti Rangi Taiao Management Plan

The Ngāti Rangi Taiao Management Plan sets out Ngāti Rangi's vision and expectations for the management of their environment. Poorly designed and maintained instream structures are identified as a critical issue impairing native fish species in their rohe. The plan sets out a specific objective that "Culverts, weirs and dams allow for native fish migration, but block trout access to uninvaded areas" (Ngāti Rangi Trust 2014). Achievement of this objective is supported by policy 4.7.1 that states "All culverts and other structures are modified or designed to ensure that no disruption to the migratory path of native fish species occurs" (Ngāti Rangi Trust 2014).

Waitaki Iwi Management Plan

The Waitaki Iwi Management Plan was developed in partnership between Te Rūnanga o Arowhenua, Te Rūnanga o Waihao and Te Rūnanga o Moeraki. It sets out a policy framework for the protection and enhancement of the Waitaki catchment. Section 6.9 identifies instream barriers as an issue for sustaining the life cycle of native fishes and sets an objective that "Mahika kai species have passage at all times" (Te Rūnanga o Arowhenua et al. 2019). This will be achieved via policies that require that "agencies develop and implement a programme to remove barriers to fish passage across the catchment" and "infrastructure (new and existing) provides for fish passage and connectivity" (Te Rūnanga o Arowhenua et al. 2019). Furthermore, the plan encourages agencies to "consider the use of structures that impede fish passage" to prevent salmonids from impacting mahika kai species (Te Rūnanga o Arowhenua et al. 2019).

Mahaanui Iwi Management Plan

The Mahaanui Iwi Management Plan sets out a policy framework intended to protect and enhance the environment and natural resources of six Papatipu Rūnanga of Ngāi Tahu. The plan promotes a ki uta ki tai approach to the management of fresh waters and highlights the need to require that "any structure…in the bed or margin of a waterway…supports and enables fish passage for migratory indigenous fish species" (WM12.13; Ngāi Tūāhuriri Rūnanga et al. 2013). The plan also recognises the need to provide sufficient water to allow for the natural passage of migratory species (WM8.2(h)).

Te Tau Ihu Mahi Tuna (Eel Management Plan)

Te Tau Ihu Mahi Tuna articulates a vision to "ensure the sustainability of the eel fishery through good management which provides for a customary, recreational and commercial harvest" (Te Tau Ihu Mahi Tuna 1999). The plan explicitly identifies obstructions to migration as a critical concern to iwi, including at irrigation infrastructure, dams, and weirs, culverts, and flood gates. One of its key recommendations is that "remedial action should be undertaken to ensure that obstacles to eel movements…are provided with suitable fish passages" (Te Tau Ihu Mahi Tuna 1999).

3.2 Performance measures and standards

Performance measures describe the metrics or variables used to measure the performance of a structure while **performance standards** describe the actual value of a metric that is needed to meet objectives (O'Connor et al. 2022). Performance measures and standards must, therefore, be aligned with and informed by the objectives.

At the scale of an individual structure, performance measures will generally align with the means objectives. Typically, this will result in both hydraulic/physical and biological performance measures (Table 3-1). **Hydraulic/physical performance measures** describe the hydraulic or physical conditions (e.g., water velocity, water depth, turbulence) within/at the structure. The hydraulic or physical conditions within/at the structure will determine how easy it is for a given fish to pass the structure and so are an important aspect of the design criteria for a structure. **Hydraulic/physical performance standards** define the acceptable range or limits for different hydraulic or physical performance measures. The design criteria for the structure must ensure that the hydraulic or physical performance standards will be achieved. Hydraulic or physical performance standards should be informed by laboratory or field-based tests of fish behaviour and passage success under different (ideally controlled or experimentally manipulated) hydraulic or physical conditions.

Biological performance measures will be derived from the biological objectives and describe different measures of fish passage success (Table 3-1). Depending on the structure type there may be biological performance measures relating to the attraction efficiency, entrance efficiency, and/or passage efficiency at the structure (Wilkes et al. 2018c). Attraction efficiency relates to the ease with which fish can find the entrance to a structure. Entrance efficiency describes the success of fish entering the structure. Passage efficiency describes the success of fish passing the structure after entering. Biological performance measures are often framed around the proportion of a species (or multiple species) successfully passing and/or delays in fish movement resulting from the structure, as these are two key metrics that are typically related to achievement of the fundamental objectives.

Biological performance measures may be established for individual species or life stages, or for the overall fish community. This will be dictated by the values that have informed the objectives. In most cases, more than one biological performance measure will be relevant for achieving the overarching fundamental objectives and this should be accounted for in the structure design and evaluation stages. **Biological performance standards** define the acceptable range or target for different biological performance measures. They should be set at a level to achieve the fundamental objectives and will typically be quantitative. Setting biological performance standards can be challenging, but as far as possible should be evidence-based and acknowledge uncertainty. They may be most effectively implemented within an adaptive management framework and tied to outcome monitoring of fundamental objectives.

Hydraulic and biological performance measures and standards may be relevant to evaluating success and should be the basis of designing effective monitoring and evaluation frameworks. For further details on relating performance measures and standards to outcome monitoring, please see Section 8 and Baker et al. (2024a).

Table 3-1: Examples of some possible hydraulic, physical, and biological performance measures that could be identified to inform structure design and evaluate success.

	Performance measure	Explanation
Hydraulic	Maximum water velocity	High water velocities may exceed the burst swimming capabilities of fish and prevent them from passing.
	Mean water velocity	If mean water velocities are high fish may become exhausted before they reach the end of the structure.
	Minimum water depth	If water is too shallow, fish cannot swim past.
	Maximum pool turbulence	Excessive turbulence can disorientate and tire fish.
	Maximum head loss	The head loss between pools in a fishway controls maximum water velocities and/or may exceed fish jumping capabilities.
Physical	Minimum fall height	If fall height is not great enough it may allow passage of undesirable species at an exclusion barrier.
	Minimum overhang distance	If the overhang distance is too short at an exclusion barrier, undesirable species may be able to jump or climb past.
Biological	Percentage passage of target species	May be defined for individual species or overall fish communities. The lower the percentage passage rate, the greater the risk of adverse effects on fish communities.
	Length of delay	Delays increase the risk of predation and may prevent fish from reaching critical habitats.
	Number of fish species passing successfully	Different species have different requirements for sustaining successful passage. Ensuring multiple species can pass may be an important measure of success.
	Size range of fish passing successfully	Individuals of different sizes have different movement capabilities. We should generally be seeking to ensure that fish of all sizes are able to pass a structure.
	Undesirable species cannot pass	In some cases, we may wish to prevent undesirable species or life stages from reaching critical habitats of threatened species.

Design of new and replacement instream structures

4 Design of new and replacement instream structures

All instream structures have the potential to adversely affect aquatic habitats and stream biota, but careful and considered evidence-based planning and design can minimise these potential impacts. The objective of the following sections is to set out recommendations and guidelines using best available information that will allow practitioners to design, install, and manage new and replacement instream infrastructure for fish passage more effectively. The intended outcome is to ensure fish passage design requirements are an integral part of the design process for instream infrastructure in New Zealand.

Design of instream structures that provide effective fish passage requires biological knowledge of fish ecology, behaviour, and the capacity of different fish species to negotiate various hydraulic conditions (e.g., velocity and turbulence), combined with hydraulic and civil engineering knowledge and expertise. This will allow development of structures that provide appropriate hydraulic conditions for fish passage, while also fulfilling requirements for hydraulic capacity and operation.

A critical challenge for practitioners and managers is accounting for the significant variations that occur between sites in fish communities, species, sizes, behaviour, and swimming abilities. Designing for fish passage requires that suitable hydraulic conditions that accommodate the different swimming capacities and behaviours of relevant fish species passing upstream and downstream at a site are provided at the appropriate design flow rates in the waterway during key migration periods.

Design approaches founded on the principles of stream simulation have now become the international standard for good fish passage design. The stream simulation design philosophy is built on the premise that mimicking natural stream conditions within the structure design should mean the structure will present no more of an obstacle to aquatic animals than the adjacent stream channel under typical flow conditions (U.S. Department of Agriculture 2008; Barnard et al. 2015). This has proven to be more effective at catering for the diverse requirements of multi-species assemblages than traditional design approaches that attempted to match hydraulic conditions within or across a structure with knowledge of specific swimming capabilities of individual target species or life stages. Furthermore, there is evidence to indicate that stream simulation designs are more resilient to extreme events and can have lower whole-of-life costs (Gillespie et al. 2014).

All new and replacement instream structures must be designed to fulfil the NPS-FM s3.26(1) requirement that the passage of fish is maintained or improved by instream structures *and* fish passage objectives set out in regional plans and fish passage action plans (see Section 2 for further details on legislative and regulatory requirements).

4.1 Principles of good fish passage design

Good fish passage design for instream structures seeks to achieve the following general objectives:

- The structure provides no greater impediment to fish movements than adjacent natural stream reaches.
- Efficient and safe upstream and downstream passage of all aquatic organisms, throughout the range of life stages, with minimal delay or injury.

- A diversity of physical and hydraulic conditions is provided leading to a high diversity of passage opportunities.
- Continuity of geomorphic processes such as the movement of sediment and debris are provided for.
- Structures are durable and have minimal maintenance requirements.

These objectives can be achieved by seeking to realise the following principles of good fish passage design:

- Maintaining continuity of instream habitat.
- Minimising alterations to natural stream alignment.
- Minimising alterations to natural stream gradient.
- Maintaining water velocities that allow for the upstream passage of native fish.
- Ensuring minimum water depths that allow for the upstream passage of native fish.
- Avoiding constraints on bank-full channel capacity resulting from the structure.
- Avoiding vertical drops.
- Providing an uninterrupted pathway along the bed of the structure.

The following sections set out good practice workflows and design principles for providing fish passage at the most common low-head instream structures. For information on fish passage design for dams, please refer to Section 7.

4.2 Design process

All sites are unique, and a case-by-case approach will be required to design instream structures to meet site-specific fish passage requirements. A general design process for instream structures is set out in Figure 4-1. It is important that an interdisciplinary team including ecologists and engineers are involved at all stages of the design process.

Initial assessment

The initial assessment phase involves collating existing catchment biological and physical information as background for defining objectives and setting performance standards for the structure. This may include an initial site reconnaissance visit to identify site-specific challenges or risks that should be accounted for in the subsequent design phases. Such factors might include locations where the stream channel is unstable laterally and/or longitudinally, places with high bed or debris loads, reaches subject to natural hazards, locations with critical infrastructure, or sites with high instream values. This stage should also include review of all relevant legislative requirements for the structure to determine what approvals or consents are required (Section 2) and an evaluation of the species present.

Defining objectives and performance standards

The information compiled during the initial assessment should be used to define ecological objectives and performance standards for the structure. Setting clear, well-defined objectives and performance standards is an important component of the design process, particularly for complex or highly valued sites, as it provides the basis for determining appropriate design criteria and for measuring and evaluating project success (see Section 3 for more details on setting objectives and performance standards).

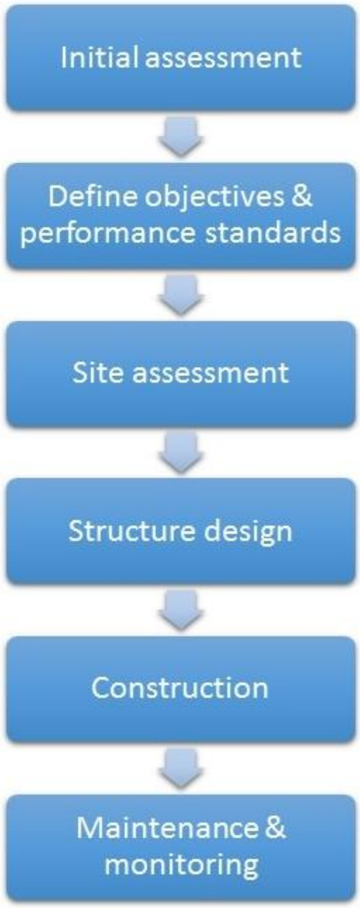


Figure 4-1: General design process for instream structures.

Site assessment and structure design

Once the objectives and performance standards are set, a site assessment is carried out to provide the reference for structure design. A design concept that fulfils the objectives and performance standards is then developed and, subsequently, final structural design drawings and specifications are prepared before the consenting and construction phase. Guidance and minimum design standards for new instream structures are set out in the following parts of this section.

Construction, maintenance, and monitoring

It is strongly recommended that relevant specialists, including a recognised fish ecologist/biologist, are present during the construction phase for more complex or high value sites to assist with responding to any unforeseen site conditions that may necessitate a deviation from the final design.

It is important to consider timing of works when planning the construction of any instream structure. Most regional plans will have constraints on the timing of instream works intended to minimise potential impacts on the aquatic environment and species. This may include avoiding works during fish migration and/or spawning periods. Please consult local plan rules to ensure compliance with these requirements. A summary of some of the key migration times of freshwater fishes in New Zealand is provided in Figure A-3.

Consideration must also be given to the practicalities of undertaking instream works. This may include requirements for redirecting stream flows during works, fish recovery and rescue, sediment control, or the appropriate use of machinery in or adjacent to waterways. Refer to Ministry for the Environment (2021) for guidance on good practice. Health and safety obligations must also be addressed.

Monitoring is required for two primary purposes:

1. to evaluate whether the structure is meeting the specified objectives and performance standards (see Section 8 for more details on this aspect), and

2. to check that the structure remains in good condition and functioning as intended, or whether maintenance is required.

Regular maintenance is essential to preserve the hydraulic and ecological functionality of a structure and/or associated fish pass. There is also a legal requirement under the FFR83 to maintain instream structures so that they continue to provide fish passage (see Section 2.1). Furthermore, failure to ensure that effective fish passage is maintained will often result in structures becoming non-compliant with regional planning rules that require fish passage.

Over time all structures will collect debris that can alter the hydraulic conditions throughout the structure and potentially create a physical or behavioural barrier to fish movements. For example, where spoiler baffles are installed, waterborne debris or large bedload movements can build up between the baffles reducing their efficacy in reducing water velocities and providing physical resting areas for fish. It is anticipated that sediment deposition will be transient and removed in subsequent flood waters, but stubborn debris may require physical removal.

Structural damage can also occur as fabrics deteriorate and components become damaged in flood flows. Artificial substrates such as spat ropes and spoiler baffle sheets can be prone to flood damage if they are not effectively fastened to the instream structure. Poorly designed rock ramps can also result in erosion of the streambed and/or the displacement of rocks during flood waters.

Development of a maintenance programme will be site- and structure-specific, but it should be focused on the migration period when the structure needs to pass fish. Several factors will determine the appropriate frequency of inspections including the type of structure, the location in the catchment, the hydrology of the river, geology of the catchment and mobility of sediments, and the type of marginal, emergent and submerged vegetation within the stream. It is advisable to develop a risk assessment matrix based on site-specific factors to inform a suitable inspection and maintenance schedule.

Monitoring and maintenance requirements over the lifetime of the structure should be considered from the outset of the design process. The higher initial construction costs of more complex designs or larger structures can sometimes be balanced by lower long-term monitoring and maintenance requirements as they provide greater capacity and resilience to extreme events. Cost minimisation at the construction phase, e.g., using smaller culverts that constrain the channel, can contribute to accelerating downstream erosion and scouring that over time results in perching and undercutting of the culvert outlet, creating fish migration barriers. This may increase maintenance requirements and future compliance costs.

4.3 Thinking like a fish for instream structure design

The objective is to ensure that most fish that arrive at a structure will pass through/over unhindered and without delay. Achieving this requires designers to “think like a fish” (sensu Williams et al. 2012). When migrating through a natural stream reach, there are a diversity of pathways that fish can choose between as they move upstream and downstream. Different species will choose different pathways; for example pelagic species such as īnanga (*Galaxias maculatus*) may select a pathway through the middle of the water column, but more towards the stream edges, while benthic species such as common bullies will move along the stream bed. Conditions in the stream also vary as flows change over time. Fish respond by altering their pathways and seeking out the path of least resistance. When designing instream structures, being aware of these differences in how fish move and interact with their environment is important. Appendix A and Appendix B provide more context on ecological considerations for fish passage design and what creates a barrier to fish, respectively.

4.4 River crossing design

River crossings are one of the most frequently encountered low-head instream structures in New Zealand. Inappropriate design and maintenance of river crossings can significantly impede fish movements. This primarily occurs when structures constrict waterways and fail to maintain continuity of natural stream habitats. Figure 4-2 summarises the suitability of a range of commonly encountered river crossing types for providing fish passage.

Single-span bridges are generally the preferred crossing type from a fish passage perspective, followed by stream simulation culvert designs. This is because these crossing types are best at maintaining the stream conditions to which fish are adapted for moving in. Single and multi-barrel box culverts are generally the next best solution and if well-designed can achieve passage efficiency similar to bridges or full stream simulation designs. Fords should generally be avoided from the perspective of catering for unimpeded fish passage. However, it is important to note that the preference will be context-specific and poor design or maintenance of any of these designs could create fish barriers.

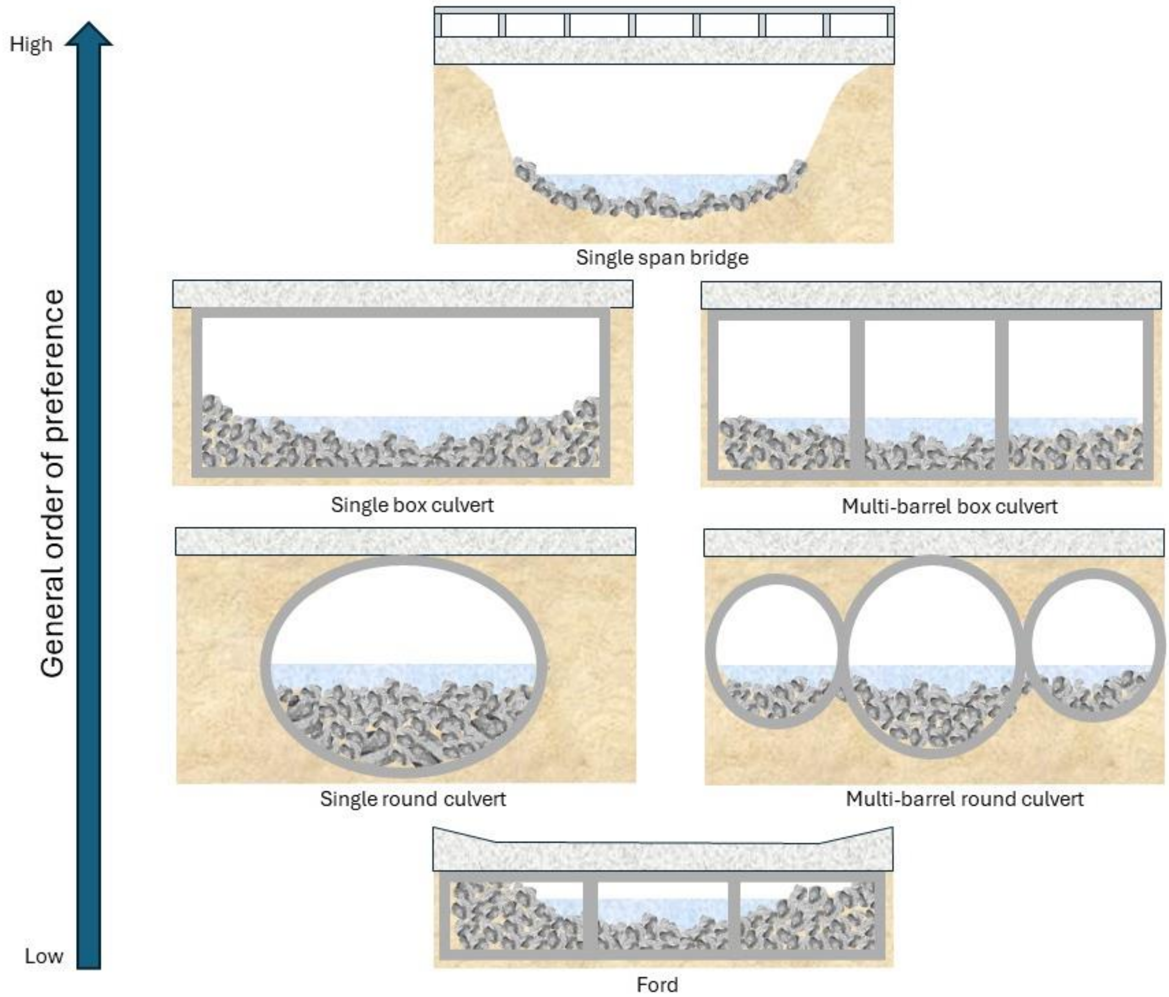


Figure 4-2: Order of preference for road crossing design based on the degree of connectivity each design facilitates.

4.5 Culverts

4.5.1 Overview

Culverts must not impede the passage of fish without approval from DOC (FFR83 s42(1)). Consequently, it is essential that all new and replacement culverts are designed to incorporate and provide for fish passage on an ongoing basis from the outset of the design process. National Environmental Standards also apply for culverts to achieve permitted activity status (NES-F R70(2); see Section 2.4.3) and **it is a requirement that the relevant regional council is notified of all new culvert installations, replacements or modifications and provided with the required information set out under the NES-F R62**.

The purpose of this section is to provide practical guidelines for the design of new or replacement culverts to ensure that fish passage is provided. These guidelines are intended to serve as a supplemental resource to qualified engineers, rather than a comprehensive manual dedicated solely to culvert design. The guidelines emphasise the importance of addressing both hydraulic requirements and fish passage objectives.

Where practicable, the guidelines incorporate reliable methods and software applications that have proven to be accurate for most streams and rivers in New Zealand. The established methods enable qualified engineers to determine how fish passage objectives can be met in culverts. By utilising these methods, engineers can ensure that the culvert design adequately considers the design flow and provides optimal conditions for fish passage.

Three general approaches for accommodating fish passage in culverts are covered within this section of the guidelines:

1. A basic method intended for culverts in small, low gradient streams or drains, where the road or access embankment does not effectively modify the floodplain.
2. A standard approach that will be applied for most culverts, especially where larger streams or higher embankments are involved.
3. A brief description of an approach for culverts in steeper terrain, especially when the culvert crossings will be at higher elevations.

To further enhance understanding and practical application, the guidelines include four worked examples (Appendix C to Appendix F). These examples demonstrate the standard approach explained in the guidelines and will allow engineers to follow the practical implementation of the guidelines in real-world scenarios. These worked examples serve as a learning tool and illustrate how the guidelines can be applied to different situations, as well as providing further clarity and guidance for culvert design.

4.5.2 Culvert design: Basic approach for small stream crossings

The basic approach for small stream or drain crossings is intended to provide a pragmatic solution for ensuring that low-risk structures provide effective fish passage. It is **only applicable for culverts that are to be placed in small, non-eroding streams or drains** (see definitions below). The basic approach is specifically **not appropriate** for:

- Under public roads, highways, or railway embankments.
- Waterways that appear to be unstable. This includes streams or waterways that are incising (bed degradation) or exhibiting bank degradation.
- Streams that are braided or anabranched (multiple interconnected channels).

Small, non-eroding streams or drains are defined for the purposes of applying the basic approach as having all the following characteristics:

- A longitudinal grade of ≤0.4% (0.04 m of vertical difference across a 10 m channel length). This grade limit must be met across a 100 m distance centred on the culvert location (i.e., from 50 m downstream to 50 m upstream of the culvert).
- A bank-full width (distance between the top of the left and right banks and see Section 4.5.4) of ≤2 m.
- A bed consisting of sand, silt, clay, or a mixture of sand, silt, and/or clay.
- For streams:

   - The driving surface or top of the road or accessway crossing the stream will be <1.8 m above the stream bed and ≤0.5 m above the tops of the stream banks and/or the adjacent ground.
   - The stream flow exceeding the 2-year ARI (annual recurrence interval) flow overtops the crossing.
- For drains:
   - The bottom/bed of the drain will be ≤1.2 m wide.
   - The drain will be ≤1.5 m deep measured from the bed to the top of the bank.
   - The drain must not have a defined catchment that extends beyond the drained land or capture off-site flow that exceeds 10% of the bank full capacity of the drain.

For the purposes of these guidelines, a drain cannot have a natural inflow at the upstream end, and discharge into a natural stream or river at the downstream end. For example, if the waterway is the only link between a natural spring and a stream or river, then it is considered a stream. In the case where a stream enters a drained section of land, the most direct connection between the stream entering and the connection to the downstream stream, river, lake, or other natural water body would be considered a stream (unless evidence indicates a different path). Larger drained areas may include tributaries to the main identified stream. If the tributaries entering the drained land provide viable fish habitat, then they must also be classified as streams.

Where the above criteria are all fulfilled, the basic approach culvert design criteria are as follows (all apply):

- The culvert must span a minimum of the bank-full width (i.e., the distance between the top of the left and right banks of the stream) where it intersects with the bed of the stream or drain, or 1.3 × the bank-full width to meet the NES-F permitted activity status (NES-F R70(2)). Where the smaller span is used, a consent will be required for installation.
- The culvert must be placed at the same grade as the stream or drain bed.
- The culvert invert must be embedded by:
   - 33–50% of the nominal diameter of a round culvert, or
   - The greater of 300 mm or 2 × D_{50} for a box culvert (where D_{50} is the median substrate size).
- The width of the embedment within the culvert must be at least as wide as the streambed, defined as the distance between the limits of terrestrial vegetation on either side of the stream.
- The maximum length of the culvert will be 6 m.
- The nominal diameter of a round culvert must be ≤1.5 m.
- The inside dimensions of a box culvert must be ≤2 m wide and ≤1.5 m high.

For streams, the intent of these design criteria is for the culvert to convey the bank-full flow at water velocities that are equivalent to the bank-full water velocities in the existing stream. In drains, the intent is to size the culvert such that it provides the same cross-sectional area, not including embedment within the culvert, as the bank-full drain and conveys the bank-full flow without head loss or upstream flooding. Generally, farm drains run at very flat grades and low water velocities, when full. By providing the same capacity as the full drain, water velocities within the culvert will be very similar to the water velocities within the drain. Furthermore, drains do not convey flow outside of their banks and often flow through floodplains in directions different to the flood flow direction within the river or stream flood flows.

4.5.3 Culvert design: Standard stream crossings

Design process

The design procedure set out in this section will be the standard approach for most stream crossings including under public roads, highways, and rail embankments. The flowchart in Figure 4-3 provides an overview of the design procedure. **A critical element of the design procedure is the need to ensure that the hydrological requirements for the site address both the design flood flows and fish passage flows.**

After setting objectives and performance standards, the design process starts with the culvert size and grade being determined for the design flows according to the requirements of the Waka Kotahi NZ Transport Agency or guidelines/standards of the appropriate local authority (section 4.5.4). The correct embedment design is then determined (section 4.5.7). Subsequently, the water velocity and depth within the culvert are tested using the upper and lower flows for fish passage design to determine whether the design satisfies the fish passage criteria (sections 4.5.5 and 4.5.6). If the design does not satisfy the fish passage criteria, the designer must adjust the size, grade, length, or number of barrels of the culvert to achieve the fish passage objectives.

Design flows

The design flows are used to determine the initial size and characteristics of a culvert, considering the site-specific flood conditions. Flood recurrence intervals are typically specified at 100 years and 10 years according to the requirements of the Waka Kotahi NZ Transport Agency or guidelines/standards of the appropriate local authority. See Section 4.5.4 for further details on hydrological methods to determine the design flows.

Fish passage flows

The design flow for fish passage is based on the bank-full flow of the stream that the culvert will be placed in. The bank-full flow is defined as the upper flow threshold (Q_H) and ⅓ of the bank-full flow is defined as the lower flow threshold (Q_L) for verifying fish passage in the culvert. A discussion of determining the bank-full flow and associated references can be found in Section 4.5.4.

Determining bank-full flow is not always practical when the stream is incised, unstable, modified by urban/agricultural land use, braided, anabranched, or the catchment hydrology has been significantly modified by land use. Under these conditions, a hydrological assessment of the contributing catchment should be completed to determine the design flows for the culvert. Generally, this is required for the design of culverts to be placed within a stream or river.

Fish passage design will require determination of the 2-year annual recurrence interval (ARI) flow (with and without the effects of climate change accounted for). Half of the 2-year ARI flow can be used as an approximation of the bank-full flow (Q_H). This closely aligns with the 1.5-year ARI flow identified by Lagasse et al. (2012). If the base flow of the stream is known, it can be used as the fish passage lower flow threshold (Q_L). However, if the base flow is unknown, or the stream is ephemeral, it is recommended to use one-tenth of the 2-year ARI flow, which is roughly equivalent to the 0.5-year ARI flow or ⅓ of the bank-full flow, for the fish passage lower flow threshold (Q_L).

Section 4.5.4 provides more detail on the hydrological methods required to determine these thresholds.

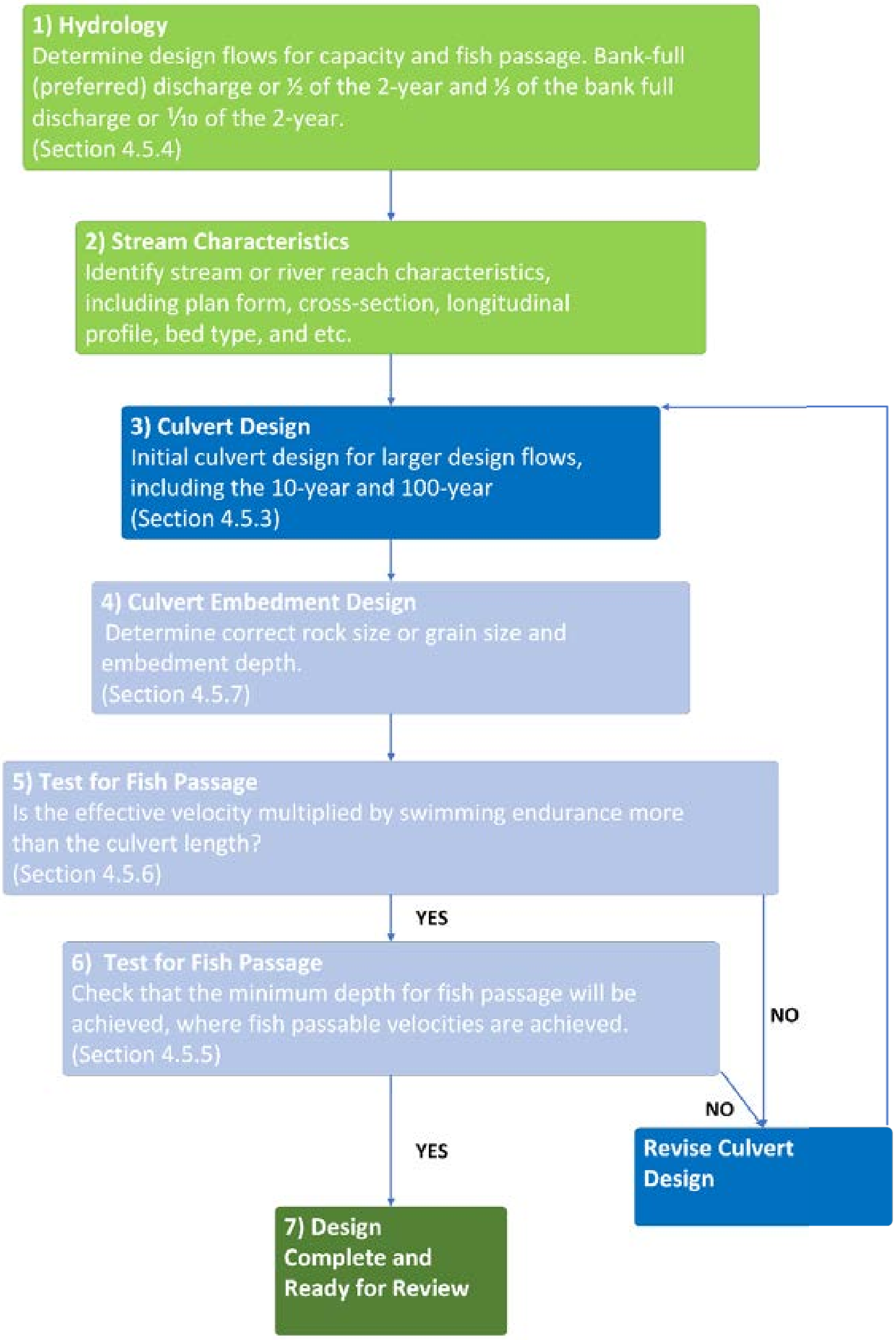


Figure 4-3: Overview of standard culvert design procedure.

Fish passage criteria

In addition to the fish passage design flows, fish passage water depth and water velocity thresholds are to be determined and defined as hydraulic performance standards. These thresholds will include maximum allowable water velocities based on the swimming capabilities of the target or benchmark species (Section 4.5.6), and a minimum water depth (Section 4.5.5).

Water velocity thresholds

Fish swimming velocity, and the duration that the swimming velocity can be maintained by the target fish species, must both be considered to determine if a culvert is passable by fish. For more background on fish swimming capabilities see Appendix A. Details on how to utilise information on fish swimming speeds in culvert design and validation are provided in Section 4.5.6.

Culvert water velocities are typically calculated as mean cross-sectional water velocities and these mean velocities often exceed fish swimming velocities and endurance. Consequently, the water velocity distribution within the culvert must be taken into consideration in determining whether suitable conditions are available for fish to successfully pass. The best practical way to account for the velocity distribution within the culvert is to look at depth-averaged vertical slices within the culvert at the fish passage design flow (Zhai et al. 2014). This approach provides a systematic methodology for determining the depth-averaged velocity of vertical slices of the flow within the culvert. Figure 4-4 shows examples of vertical slices over a typical velocity distribution within a rectangular channel. The depth-averaged velocity within the vertical slice outlined in red would be much lower than the vertical slice in yellow. The vertical slice outlined in red would also be significantly less than the mean cross-sectional water velocity. Section 4.5.6 provides details on methods to determine water velocities within depth-averaged vertical slices for fish passage design.

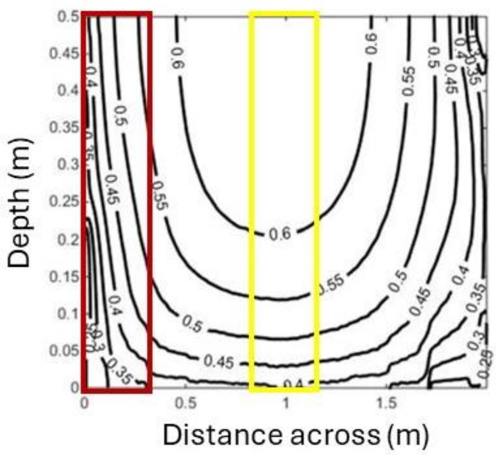


Figure 4-4: Cross-sectional water velocity distribution within a rectangular channel with example vertical slices. Modified from Cassan et al. (2019)

Minimum water depth

The culvert conveying the fish passage design flow should have a minimum water depth of 150 mm at Q_L for native species. Where passage of Salmonidae (salmon and trout) is to be accommodated, a minimum water depth of 250 mm is necessary at Q_L. The minimum water depths must be provided within areas of the culvert that also provide water velocities within the fish passage water velocity thresholds (see below). See Section 4.5.5 for further details.

Embedment design

The culvert invert should be embedded within the stream bed to facilitate fish passage. Round culverts should have a minimum embedment that is ⅓ to ½ of the culvert diameter. Minimum embedment height for box culverts should be the greater of 300 mm or 2 × D_{50} (where D_{50} = median substrate size).

For bottomless culverts, or culverts with full stream simulation, the shear stress method should be applied to determine D_{50}. It is assumed that the bed particle initiates movement when the average shear stress exceeds the critical shear stress. See Section 4.5.7 for further details. Given the scour patterns in bottomless culverts and stream simulation. The embedment design shall be based on the maximum applied shear stress on the channel bed and the equation formulated by Kerenyi and Pagan-Ortiz (2007) to determine rip-rap size, which considers the local velocity at the culvert entrance (see Section 4.5.7).

A discussion of determining the bottomless culvert aprons can be found in section 4.5.7 and in the worked example in Appendix F.

Summary design guidance for standard stream crossings

- ☐ Fish passage upper (Q_H) and lower (Q_L) flows should be defined. Q_H is bank-full discharge and Q_L is 1/3 of bank-full discharge. If the stream or river is incised, unstable, or has been modified by urban or agricultural land use, if it is braided or has anabranched, or if the catchment hydrology has been altered, Q_H is half of the 2-year flow and Q_L is one-tenth of the 2-year flow.
- ☐ Prolonged swimming speed is utilised for the target species of NZ native fish, īnanga.
- ☐ The 75% quantile prolonged swimming speed data are used to obtain swimming endurance (s).
- ☐ Fish swimming distance, D_s (m), is determined for Q_H by evaluating (Prolonged Swimming Speed – Culvert Velocity) × Swimming Endurance
- ☐ Fish passage criteria will be:
  - Fish swimming distance should be more than the culvert length at Q_H
  - Depth of flow through the culvert should be more than 150 (mm) for target native species, and 250 (mm) for Salmonidae at Q_L
- ☐ Embedment criteria will be:
  - For round culverts, a minimum embedment is between 1/3 and 1/2 of the culvert diameter
  - For box culverts, the minimum embedment should be either 300 (mm) or 2 × D_{50}, whichever is greater
  - Embedment bed material should be stable in 100-year and 10-year ARI flows.
- ☐ Ensure ancillary structures (e.g. aprons) also meet fish passage criteria.

4.5.4 Hydrology: Design & fish passage flows

Culvert design requires identifying both the design flood flows (typically 100-year and 10-year flows) and the fish passage design flows (Q_H and Q_L). The design flood flows will be determined using flow estimation methodologies; the Soil Conservation Service (SCS) method and Storm Water Management Model (SWMM) are outlined below for this purpose. Where practicable, the fish passage design flows are established based on determining the magnitude of bank-full flows. However, assessing the bank-full flow can be challenging, particularly when the stream channel has undergone modifications or incisions caused by urbanisation, agriculture, or other land use activities that affect hydrology. In such circumstances, hydrological flow estimation methods can be used to determine the 2-year ARI flow, with ½ of the 2-year ARI used as an approximation of the bank-full flow.

In the case of multi-barrel (cell) culverts, the assessment process becomes more complex. If the multi-cell culvert has been designed for fish passage to occur during low flows in the primary, and high flows in the side culvert(s), then it must be confirmed that there is a flow rate for which both the primary and side culvert(s) yield suitable fish passage conditions.

Bank-full flow determination

Bank-full flow is the maximum flow that a channel can convey without overflowing onto the floodplain, as depicted Figure 4-5. An alternative common definition of bank-full stage is the level where the width to depth ratio is a minimum as shown in Figure 4-6 (Copeland et al. 2000). In many New Zealand streams and rivers, bank-full flow in stable channels corresponds to an annual flood recurrence interval (ARI) of approximately ½ of the 2-year flood (or the 1.5-year recurrence flood which is reasonably close to ½ of the 2-year ARI).

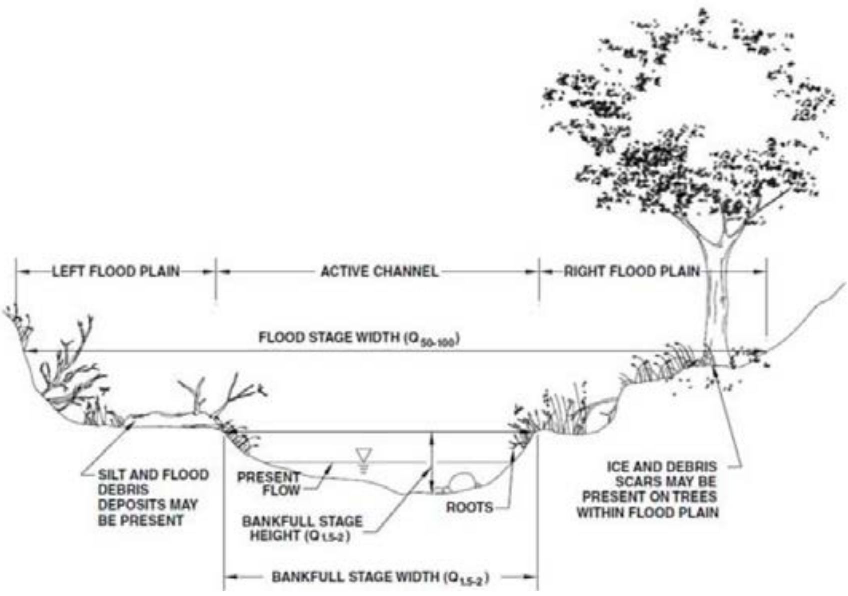


Figure 4-5: Schematic showing bank-full stage width and height.

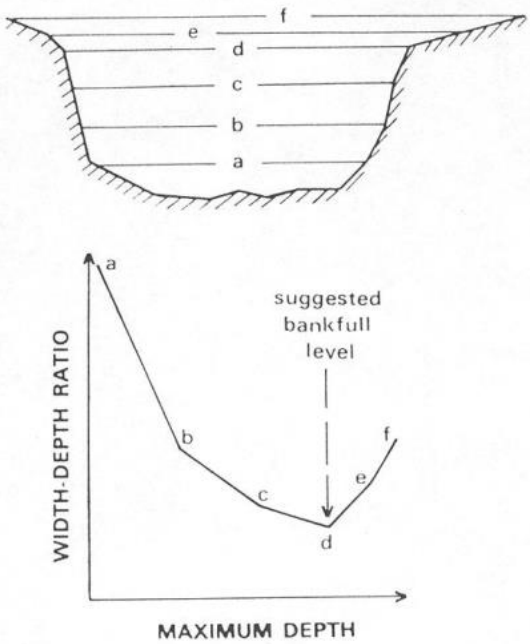


Figure 4-6: Bank-full depth using width-depth ratio. Relies on accurate field surveys, after Knighton (1998).

The process of identifying bank-full indicators in the field can be challenging and subjective. It should be conducted in stream reaches that are stable and alluvial in nature, as per the recommendation of Knighton (1998). The stream reach should be identified as stable and alluvial before field personnel attempt to identify bank-full stage indicators. In cases where the project reach is unstable or non-alluvial, alternative options include exploring stable alluvial reaches upstream or downstream within the same stream to identify indicators of bank-full stage. The process of identifying such indicators is typically iterative and heavily reliant on expert judgement.

The following guidelines are provided relative to field determination of bank-full flow:

- Bank-full flow is geomorphologically significant only in stable alluvial channels. As such, it is important to ensure that the reach where bank-full stages are measured is stable and that the stream bed remains mobile during bank-full flow.
- To determine channel dimensions for the main channel using bank-full flow, it is necessary to use top-of-bank indicators as field indicators for identifying the bank-full stage. Indicators such as the edge of the active channel, the beginning of woody vegetation, or the top of channel bars may be useful for designing specific features in a restored channel, but they should not be used to establish the bank height of a stable channel. It is only bank-full flows, which are top-of-bank flows, that are morphologically significant in establishing the bank-full flow.
- There is an exception to the rule stated above, which applies to a stable and alluvial incised stream that has formed a new floodplain within the incised channel. In this case, the top of the high bank is an abandoned floodplain or terrace, and there should be newly formed top-of-bank features within the older incised channel. However, it is important to remember that the new floodplain may not yet be fully formed, meaning that the channel may not be stable and may still be aggrading. This could result in misleading values for the bank-full flow.

- It may not be appropriate to assume that the bank-full flow for one reach of a stream is equivalent to the bank-full flow in another reach. The location of the break between the channel and the floodplain is affected by various factors, such as, confinement of the floodplain, hydrologic regime, sediment supply, bed and bank sediment size and cohesiveness, size, and type of vegetation on the floodplain and within the channel, and controls on channel width, slope, and alignment.

If a reach is not stable and alluvial, indicators of bank-full stage will be unreliable. The following examples serve to illustrate this fact:

- If a reach is non-alluvial, the sediment transport capacity typically exceeds the sediment supply, leading to the absence or underdevelopment of deposits. Utilising underdeveloped deposits as indicators of bank-full indicators would result in a flow that is too low to produce the necessary channel forming flow. Furthermore, deposits may represent relics of extreme flood events, thereby yielding a flow value that is normally too high for channel forming flow.

- In the event of channel degradation, the transport capacity of sediment exceeds the supply of sediment, resulting in the validity of the observations mentioned earlier for non-alluvial channels. Furthermore, the channel bed lowering leads to the abandonment of previous floodplain deposits, which are gradually transforming into terraces. Reliance on these features as indicators would result in an overestimation of the bank-full flow.

- On the other hand, if the channel is aggrading, in-channel deposits may be inaccurately interpreted as indicators of bank-full stage. Due to the rising bed of the stream, utilising the current floodplain as an indicator would result in an underestimation of the bank-full flow. Although the floodplain will aggrade eventually, this process typically occurs at a slower rate than that of the channel.

Flow estimation methodology

Soil Conservation Service (SCS) curve number surface runoff method

Whenever the rate of water application to the ground surface exceeds the rate of infiltration, surface runoff occurs. Initially, when water is applied to dry soil, the infiltration rate is typically high (Neitsch et al. 2011). However, as the soil gets wetter, the infiltration rate gradually reduces. As soon as the application rate exceeds the infiltration rate, surface depressions start to fill. Once all surface depressions are filled and the application rate is still higher than the infiltration rate, surface runoff will take place.

The SCS curve number equation is:

$$Q_{runoff} = \frac{(R_{24} - 0.2S)^2}{(R_{24} + 0.8S)} \tag{1}$$

Where:

Q_{runoff} = accumulated runoff or rainfall excess (mm)

R_{24} = rainfall depth for the day (mm)

S = retention parameter (mm)

The retention parameter is subject to spatial variation due to changes in soils, land use, management, and slope, as well as temporal variation due to changes in soil water content. Its definition is as follows:

$$S = 25.4 \times \left(\frac{1000}{CN} - 10\right) \quad (2)$$

Where:

CN = curve number for the day

The SCS curve number, CN, is a function of the permeability of the soil, land use, and the previous soil moisture conditions.

The following steps can be adopted to estimate the CN, from (Neitsch et al. 2011):

- Determine "Hydrologic Soil Group" from Chapter 7, Part 630 Hydrology, National Engineering Handbook, United States Department of Agriculture[7].
- Determine "The Land Uses and Treatments Group" from Chapter 8, Part 630 Hydrology, National Engineering Handbook, United States Department of Agriculture[8].
- Determine "Curve Number", CN, from Table 9-1 Chapter 9, Part 630 Hydrology, National Engineering Handbook, United States Department of Agriculture[9].
- Apply "Soil Moisture Condition Adjustments" to the CN value.

SCS uses three moisture conditions. *I*-dry (wilting point), *II*-average moisture, and *III*-wet (field capacity). In dry condition, the moisture condition *I* curve number represents the minimum value that can be assigned to the daily curve number. The equations used to calculate the curve numbers for moisture conditions *I* and *III* are as follows:

$$CN_1 = CN_2 - \frac{20 \times (100 - CN_2)}{\left(100 - CN_2 + e^{\left(2.533 - 0.0636 \times (100 - CN_2)\right)}\right)} \quad (3)$$

$$CN_3 = CN_2 \times e^{\left(0.00673 \times (100 - CN_2)\right)} \quad (4)$$

Where:

CN_1 = moisture condition *I* curve number

CN_2 = moisture condition *II* curve number

CN_3 = moisture condition *III* curve number

[7] https://directives.sc.egov.usda.gov/22526.wba
[8] https://directives.sc.egov.usda.gov/18386.wba
[9] https://directives.sc.egov.usda.gov/17758.wba

Slope adjustments

The curve numbers for moisture condition *II* listed in the tables are presumed suitable for slopes of 5%. However, Williams (1995) formulated an equation to adjust the curve number to a slope of varying degree.

$$CN_{2s} = \frac{(CN_3 - CN_2)}{3} \times \left(1 - 2 \times e^{(-13.86SL)}\right) + CN_2 \tag{5}$$

Where:

CN_{2S}	= moisture condition *II* curve number adjusted for the slope
CN_2	= moisture condition *II* curve number for the default 5% slope
CN_3	= moisture condition *III* curve number for the default 5% slope
SL	= the average slope of the sub catchment

Environmental Protection Agency Storm Water Management Model

The Environmental Protection Agency Storm Water Management Model (EPA SWMM) is an advanced software application that generates surface runoff hydrographs from sub-catchments and then routes and combines these hydrographs. This software can simulate a diverse range of hydrological processes, such as infiltration, runoff generation, and the transportation of pollutants in drainage systems. By employing the "Dynamic Wave Method", which involves applying the complete "Saint-Venant" Equations, EPA SWMM can route runoff effectively. The EPA SWMM implements Horton, modified Horton, Green Ampt, modified Green Ampt and Curve Number infiltration equations as its infiltration methods. These approaches utilise infiltration rates corresponding to diverse soil types in the sub-catchment (James et al. 2010).

SWMM model inputs: Manning`s roughness

The values of Manning's roughness coefficient (n) for the left bank, right bank, and main channel of the transect are recommended to be determined from HEC-20 and Hicks and Mason (1998). The designer may use other methods appropriate for the situation. However, care should be taken in selecting a method and consideration should be given to performing a sensitivity analysis on the Manning's n because the methods do not always provide consistent results. In the case of embedded culverts, composite roughness is applied following the Horton-Einstein equation:

$$n_{composite} = \left(\frac{P_{side}(n_{culvert})^{1.5} + P_{bottom}(n_{bottom})^{1.5}}{P_{side} + P_{bottom}}\right)^{\frac{2}{3}} \tag{6}$$

Where:

$n_{composite}$	= Mannings roughness coefficient for multiple materials
P_{side}	= Perimeter of side material
$n_{culvert}$	= Mannings roughness coefficient for culvert material

P_{bottom} = Perimeter of bottom material

n_{bottom} = Mannings roughness coefficient for bottom material

SWMM model inputs: Soil type

The infiltration rate is greatly influenced by the soil type, as it determines the soil's porosity, permeability, and clay content, all of which have a significant impact on the ability of water to infiltrate the soil surface. S-map is an online tool that covers New Zealand soil information where you can obtain detailed soil data including soil type and clay content.

SWMM model inputs: Horton Infiltration Parameters

This method is based on empirical observation that demonstrates the reduction of infiltration rate from a maximum value to a minimum value over the course of an extended rainfall event (Figure 4-7). The method requires input parameters such as the maximum and minimum infiltration rates, a decay coefficient that characterises the rate's rate of decline over time, and the duration required for a fully saturated soil to dry completely (utilised to determine the restoration of infiltration rate during dry periods).

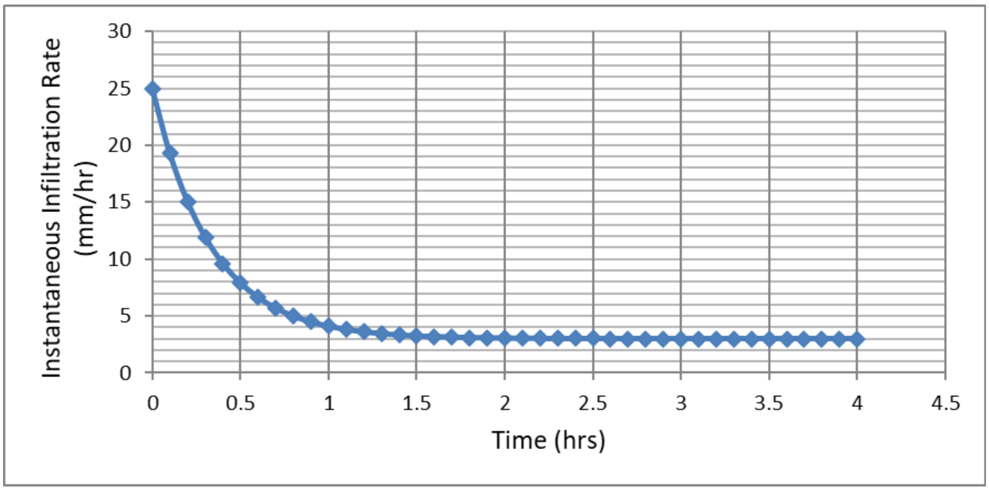


Figure 4-7: Horton infiltration rates for sub-catchments. An example of exponential decrease of infiltration rate.

4.5.5 Fish passage criteria: water depth

The depth and cross-sectional area of the fish passage design flow in a culvert should be as close to the bank-full depth and cross-sectional area in the upstream and downstream reaches of the stream as possible. However, due to the difference in effective roughness between the culvert and the stream, the depth and cross-sectional area within the culvert will vary from that of the stream.

The culvert conveying the fish passage design flow should have a minimum water depth of 150 mm at Q_L for native species. Where passage of Salmonidae (salmon and trout) is to be accommodated, a minimum water depth of 250 mm is necessary at Q_L. The minimum water depths must be provided within areas of the culvert that also provide water velocities within the fish passage water velocity thresholds (see below).

4.5.6 Fish passage criteria: water velocity

Fish swimming capabilities and culvert design

Culvert design for fish passage demands an understanding of fish swimming capabilities and how they apply to the passage of fish past obstacles in streams.

Much as humans can sustain a walking pace for far longer than a run, fish have different swimming speeds and can sustain them for different amounts of time. Fish have three basic categories of swimming speed – sustained, prolonged, and burst (Beamish 1978). Sustained swimming is fuelled aerobically and can be maintained indefinitely (theoretically), without muscle fatigue. Experimentally, it is typically defined by a swimming speed that a fish can maintain for greater than 200 minutes. The maximum sustained swimming speed is typically approximated using a critical swimming speed test (Brett 1964). However, more recent research suggests that the maximum sustained swimming speed is only 60–80% of the critical swimming speed (Burgetz et al. 1998; Richards et al. 2002; Amérand et al. 2017; Hvas and Oppedal 2017; Hvas et al. 2021). For design purposes, the maximum sustained swimming speed should, therefore, be assumed to be around 70% of a measured critical swimming speed. Critical swimming speeds for native species are given in Table 4-1. Fish use sustained swimming for activities like migration and foraging. Burst swimming is an anaerobically-fuelled process that leads to muscle fatigue over a period of less than 20–30 seconds. Fish typically use burst swimming for prey capture and predator avoidance.

Table 4-1: Mean critical swimming speeds for a range of native fishes during their primary upstream migratory life stage. Source: Crawford et al. (In review).

Species	Mean maximum sustained swimming speed ($m\ s^{-1}$)	Mean critical swimming speed ($m\ s^{-1}$)	Mean size (cm)
Banded kōkopu (*Galaxias fasciatus*)	0.25	0.36	4.2
Common bully (*Gobiomorphus cotidianus*)	0.24	0.34	3.2
Giant kōkopu (*Galaxias fasciatus*)	0.44	0.63	4.9
Īnanga (*Galaxias maculatus*)	0.20	0.28	4.4
Kōaro (*Galaxias brevipinnis*)	0.40	0.57	5.2
Longfin elver (*Anguilla dieffenbachii*)	0.22	0.32	11.0
Redfin bully (*Gobiomorhpus huttoni*)	0.29	0.41	4.1

Species	Mean maximum sustained swimming speed (m s^{-1})	Mean critical swimming speed (m s^{-1})	Mean size (cm)
Shortjaw kōkopu (*Galaxias postvectis*)	0.39	0.56	5.7
Smelt (*Retropinna retropinna*)	0.43	0.61	7.3

Prolonged swimming describes an intermediate category of speeds between the low speeds that fish can sustain indefinitely and the very high speeds that cause fatigue in a matter of seconds. In the range of prolonged swimming speeds, fish are using aerobic and anaerobic pathways and will eventually fatigue. The time to fatigue is related to the swimming speed. Experimentally we can derive a relationship between swimming speed and the time that a fish can maintain that speed before fatiguing. This relationship has been derived for a small number of New Zealand species for which there is adequate experimental data (Figure 4-8).

The design of structures for fish passage typically targets the prolonged swimming speed range. Sustained swimming speeds are often impractically slow for design, and at the other end of the scale burst speeds would not give fish enough time to pass a structure. The most important aspect of prolonged swimming to consider is that it causes fatigue over time, and swimming in the prolonged speed range can, therefore, only be maintained for a limited time. This imposes two critical design criteria – fish swimming speed *and* the time that speed can be maintained for.

We cannot, however, simply design the water speed to match the fish swimming speed. To make passage over the ground, the fish must swim faster than the water speed. It is, in fact, the difference between the swimming speed and the water speed that determines how long it takes a fish to pass through a culvert, and how long a fish must maintain a given swimming speed. Consequently, **the hydraulic design must ensure that a fish can maintain the required swimming speed for long enough to ensure it progresses upstream through a culvert against the design water speed in less time than leads to fatigue** (see next section).

Significant variability in swimming performance exists between individuals within a species. Swimming ability tends to be positively correlated with size, and random variability is still large aside from this. Traditionally, fish passage structures have been designed from average swimming speeds, but this excludes a large proportion of the population. In addition to leading to overall population reductions, this can create selection pressures that favour the strongest and largest swimmers in a population (Jones et al. 2020). Most guidance suggests that fish passage efficiency (i.e., the proportion of fish that should pass) should be >90% to sustain wider population processes (Lucas and Baras 2001; O'Connor et al. 2022).

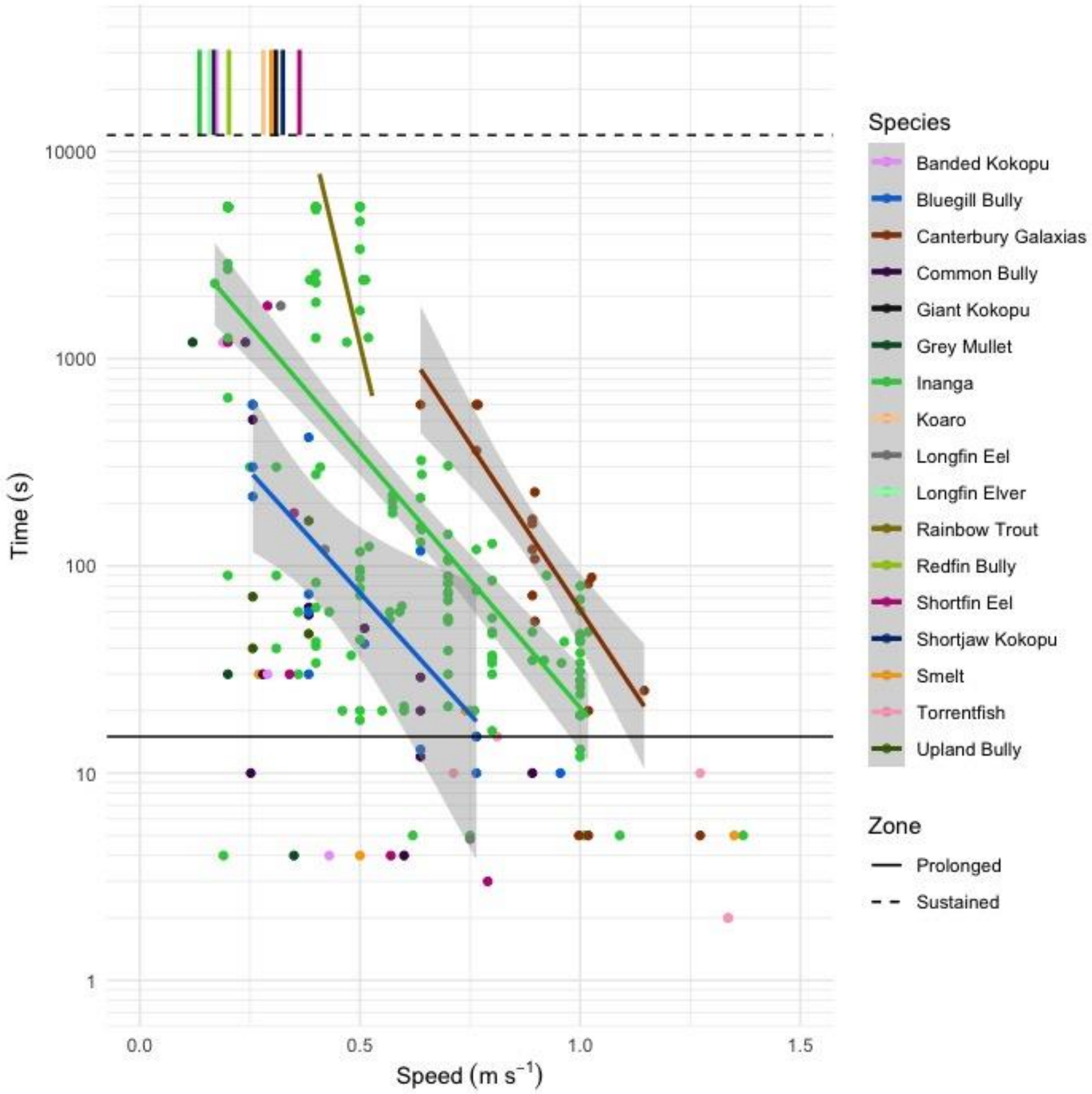


Figure 4-8: Swimming speeds and associated endurance times for various native species and for rainbow trout. The horizontal lines delineate the approximate break points between burst, prolonged and sustained swimming. Burst swimming (below the solid line) can be only sustained for several seconds. Prolonged swimming (between the solid and dashed lines) can be sustained for longer periods, but fish fatigue eventually depending on the speed. Sustained swimming (above the dashed line) can theoretically be sustained indefinitely. Points show the raw data (if available) or average data reported for each species from each source publication. Regression lines have been fitted with 95% confidence intervals for īnanga, Canterbury galaxias, and bluegill bully for data in the prolonged zone (between the solid and dashed horizontal lines). A single line represents the line of best fit for rainbow trout from the source publication (no raw data plotted). Vertical lines represent the predicted sustained swimming speed based on 70% of the average critical swimming speed for a species. Data for native species were sourced from Mitchell (1989), Boubée et al. (1999), Langdon and Collins (2000), Nikora et al. (2003), Bannon (2006), Plew et al. (2007), Tudorache et al. (2015), Nolte (2019), Crawford et al. (2023), Crawford et al. (In review), and NIWA (unpublished data). Data for Rainbow trout were sourced from Beamish (1978).

Fish swimming speeds

The fish passage water velocity threshold will be based on the measured swimming capabilities of the target or benchmark species (e.g., īnanga) identified when setting objectives. As described above, fish swimming speed can be categorised into three general ranges with respect to their physiological capabilities and limitations. Burst speeds can only be sustained for a short period of time (<20 s) and so are not suitable for defining hydraulic design criteria for culvert design. Sustained swimming speeds can theoretically be sustained for extended periods (>200 min) without fatigue. Ideally, culverts would be designed to provide a continuous pathway with water velocities that allow for passage using the sustained swimming mode. However, this can be impractical and so information on the prolonged swimming speeds of fish will typically be used as the basis for defining the fish passage water velocity threshold.

To determine whether a culvert meets the water velocity requirement for fish passage, it is essential to consider the swimming speed (m s^{-1}) of the fish within the culvert and the endurance (s) for which the target fish species can maintain that swimming speed. To pass upstream through the culvert, the fish must swim at a speed that exceeds the water velocity within the culvert, i.e., $V_f > V_c$, where V_f is the swimming speed of the fish within the culvert and V_c is the velocity of the water within the culvert that the fish is swimming through. The effective velocity (V_{ef}) of the fish passing through the culvert is determined by subtracting the culvert water velocity (V_c) from the prolonged swimming speed of the fish (V_f). Because prolonged swimming results in fatigue in the fish, it is also necessary to account for the duration for which the target or benchmark species can sustain the prolonged swimming speed in defining the fish passage water velocity threshold. The swimming distance (D_s) that the fish can achieve within the culvert is calculated by multiplying the effective velocity (V_{ef}) by the duration (t_p) for which the fish can maintain its prolonged swimming speed (see Equation (7)). The final step is to compare the fish swimming distance D_s with the length of the culvert *L*. If the fish swimming distance is greater than the culvert length ($D_s > L$), it indicates that the fish can theoretically successfully pass through the culvert.

$$(V_{fp} - V_c) \times t_p = V_{ef} \times t_p = D_s \tag{7}$$

Where:

- $V_{fp} > V_c$ and V_{fp} is the prolonged swimming speed of the fish within the culvert in metres per second and V_c is the velocity of the water within the culvert in metres per second.
- Effective velocity (V_{ef}) of the fish through the culvert in metres per second is $V_{fp} – V_c$.
- t_p is the duration that a fish can maintain a prolonged swimming speed (V_{fp}) in seconds.
- D_s is the maximum swimming distance in metres that the fish can attain at V_{ef}.

For the purposes of the standard culvert design approach in low gradient streams (≤0.4%), īnanga should be used as the benchmark species for evaluating culvert water velocities. Crawford et al. (In review) have recently quantified īnanga swimming endurance in laboratory tests (Figure 4-9). These data can be used as the basis for identifying swimming endurance (t_p) at different prolonged swimming speeds (V_{fp}). While most guidance recommends that fish passage efficiency targets should be >90% to sustain wider population processes, given the relatively low sample sizes used in Figure 4-9 (and hence high uncertainty in the 90th percentile), it is recommended that the 75th percentile endurance time (i.e., t_p that is equalled or exceeded by 75% of fish) be used as the basis for determining the fish passage velocity threshold until such a time that more data are available to refine the 90th percentile. Furthermore, to account for higher fatigue rates within the transitional zone between prolonged and burst swimming modes (i.e., between 0.7 m s^{-1} and 0.8 m s^{-1}), an adjustment is made to the culvert length (*L*), multiplying it by 1.1, before comparing it to D_s. Additionally, the integration of upstream and downstream stilling basins shall be included, to ensure that fish can successfully navigate the culvert length.

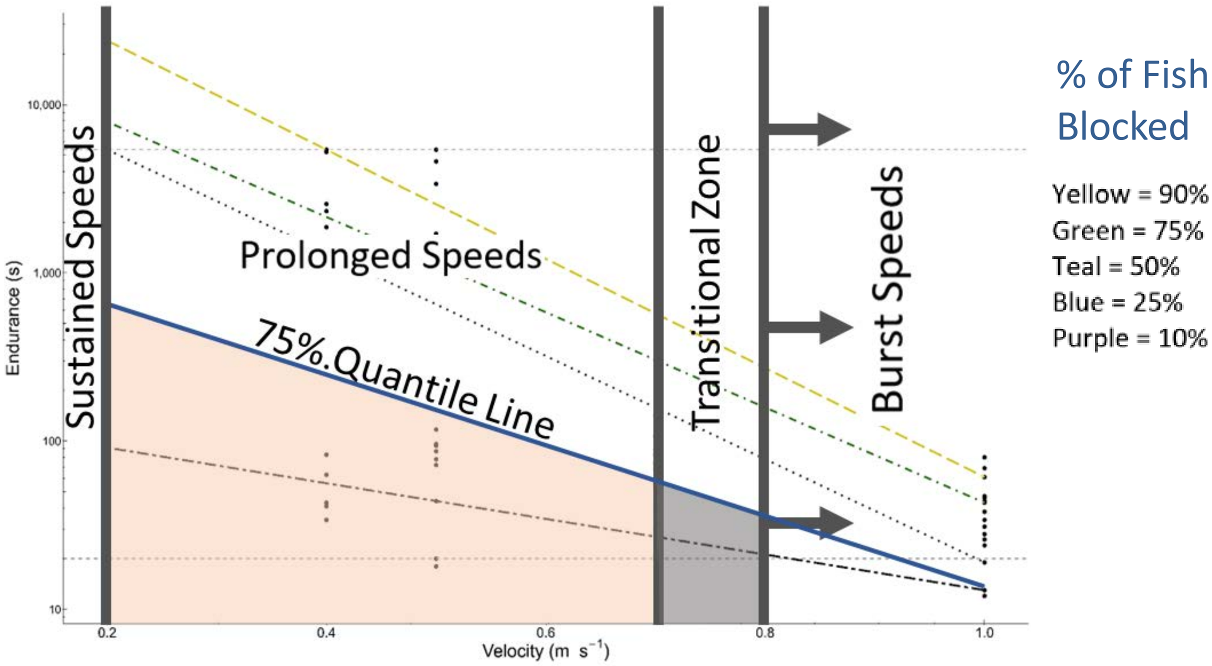


Figure 4-9: Swimming speeds and endurance for īnanga. Quantile regression was used to establish relationships between swimming speed and endurance for different percentages of individuals. Adapted from Crawford et al. (In review).

Unfortunately, using culvert mean cross-sectional water velocities will almost univerally result in a negative V_{ef} for sustained or prolonged fish swimming speeds, indicating that the fish will not be able to progress at all through the culvert. Increasing the size of the culvert enough to reduce the mean velocity below V_{fs} or V_{fp} will usually yield culverts that are impractical due to excessive size and/or excessive maintenance due to constant sediment accumulation.

The following section describes a hydraulic design approach that explicitly utilises the spatial variation in water velocities that exist within a culvert cross-section to ensure a continuous pathway exists within the culvert that is below the fish passage velocity threshold.

Depth-averaged water velocity design

Generally, the hydraulic calculations for channel or culvert water velocity (e.g., using HY-8 or HEC-RAS – see below) generate an estimated cross-sectional mean water velocity (V_{cx}). Where practicable, culverts should be designed such that V_{fs} or $V_{ps} – L/t_p > V_{cx}$ at the fish passage design flow (Q_H). However, the mean cross-sectional water velocity of the culvert (V_{cx}) will often exceed these thresholds (i.e., V_{ef} will be negative) under these conditions. Despite this, water velocities along the sides, bottom, and lower corners of a box culvert or the around the perimeter of a circular culvert will be less that the cross-sectional average (V_{cx}) and may provide a passable pathway for fish (e.g.,

Figure 4-4). In all cases, when applying the depth-averaged vertical slice hydraulic design approach, a minimum depth threshold at the low fish passage design flow (Q_L, i.e., 10% of the 2-year ARI flow) of 150 mm for target native species or 250 mm for salmonids (trout and salmon) applies within the vertical slices where the velocity meets the fish passage velocity threshold.

In order to accomodate upstream movement of native fish, the water velocity distribution within the culvert cross-section must be taken into consideration. The most effective and practical method to account for the velocity distribution is by examining cross-sectional depth-averaged slices of the design flow within the culvert, which provides a systematic methodology for determining the depth-averaged velocity of vertical slices of the flow within the culvert (Figure 4-10). Zhai et al. (2014) provides an empirically derived approach for approximating culvert velocity distribution with depth averaged velocities in vertical slices of the flow within the culvert. As illustrated in Figure 4-10, and demonstrated by Zhai et al. (2014), even when the mean cross-sectional water velocities are too high for native New Zealand fish to overcome, there are areas within the cross-section of culverts where the water velocity is below the fish passage velocity threshold. In practice, **the minimum width of the vertical slice that is fish passable must be 150 mm and the minimum depth must be achieved within the fish passable vertical slice.**

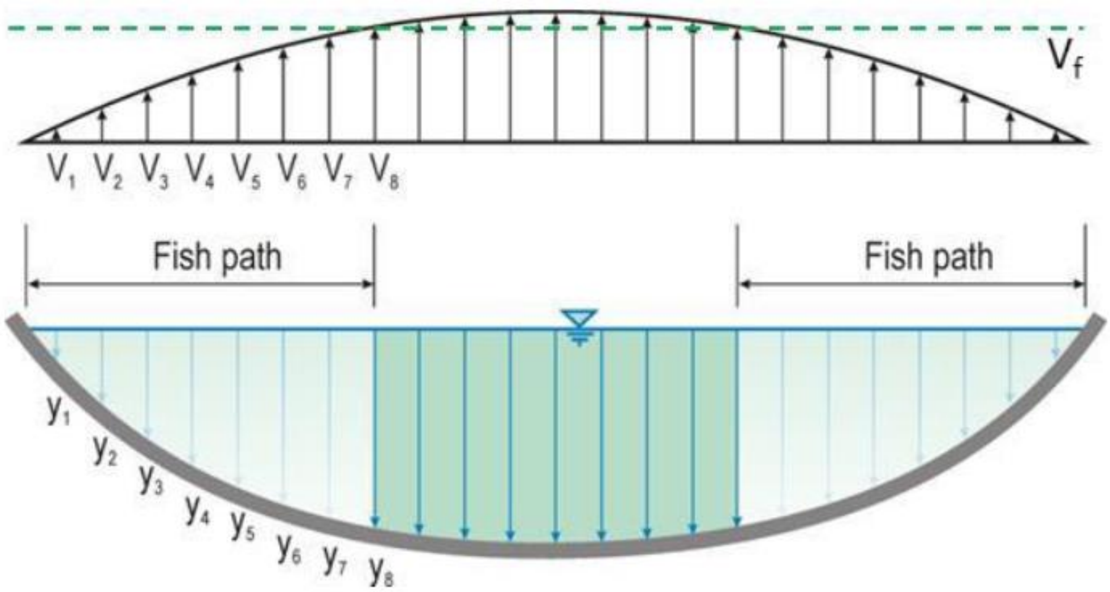


Figure 4-10: Culvert water velocity distribution with depth average velocities in vertical slices.

There are three simple and readily available approaches to account for the water velocity distribution in culvert cross-sections:

- **Utilising HY-8 software**

  HY-8 is culvert design and analysis software published by the US FHWA. HY-8 directly applies the methods for determining depth averaged water velocities in Zhai et al. (2014). It features a Low Flow option that calculates depth-averaged velocities within a vertical slice of the culvert based on the theoretical velocity distribution and the mean velocity in the culvert. This calculation is performed as a function of distance from the culvert wall. See the worked example in Appendix C for further information.

- **1D Modelling utilising HEC-RAS**

  The culvert water velocity during fish passage design flow (Q_H) can be modelled using 1D HEC-RAS. The resulting velocity is divided by three to estimate the depth-averaged velocities. This approach can only be applied where the culvert is outlet controlled and the “Froude Number” at the inlet and outlet ends is less than 0.5 when conveying the fish passage design flow. This approach will provide results that align with the method of estimating depth averaged water velocities in vertical slices of the flow within the culvert, described in Zhai et al. (2014). This method also allows the modelling of the inlet and outlet aprons' integration into stream hydraulics. This approach is demonstrated in the worked examples in Appendix D and Appendix E. Where fish passable water velocities are not achieved with this method, i.e., if V_{mean} /3 is not below the fish passage velocity threshold), the HEC-RAS input and output can be used to populate the HY-8 input and the HY-8 method described above applied.

- **HEC-RAS 1D Velocity Distribution for Bottomless Culverts and Bridges**

  HEC-RAS 1D can be used with the “Velocity Distribution” function to provide a reasonable water velocity distribution within bottomless culverts and short span bridges, or culverts with full stream simulation (refer to the worked example in Appendix F). Bottomless culverts and culverts with full stream simulation are entered into HEC-RAS 1D as bridges, which will determine water velocity distributions under the upstream and downstream edges of the bridge. When applying this method to culverts with stream simulation, it is important that design flows other than the fish passage design flow are modelled as culverts, as bridge and culvert methods may not provide the same results as flows increase relative to culvert capacity. HY-8 does not work with bottomless culverts or short span bridges and cannot take into account channel shapes within the culvert and so is not appropriate for stream simulation designs.

In some cases, the water velocities around the wetted perimeter of the culvert will be too close to the limit of native fish swimming speeds for effective passage when the flow reaches ½ of the 2-year ARI flow (i.e., Q_H). In the case of new structures, an alternative crossing design (e.g., a bridge) may be required to achieve the fish passage objectives. In the case of replacement of an existing structure where pre-existing site constraints may prevent an alternative crossing design, two further mitigation measures may help achieve passage:

- A sill combined with a depressed pool can be incorporated at the downstream end of the culvert. This will raise the culvert tailwater, increasing the depth and decreasing water velocity within the culvert during smaller flood flows. This will also provide a large resting area for native fish, prior to entering the culvert. A sill is most useful if the impact of the tailwater increase extends throughout the length of the culvert to the inlet and hence is most suitable for shallow flows, very flat longitudinal grades, and/or short culverts. **Care must be taken to ensure that the sill does not impede fish passage** (see Section 5.5.1 for more information on backwatering).

- If modelling indicates that that a sill and depressed pool will not reduce water velocities adequately, then devices (e.g., spoiler or vertical baffles) that reduce water velocities and provide resting points can be fitted into the culverts. These devices will be selected to minimise impacts to culvert maintenance and capacity. However, the interaction of these devices with the water and embedment causes a very complex flow field. It is, therefore, essential that the impacts of these devices are understood in detail, which can only be obtained through bespoke modelling. See Section 5.5.6 for details on suitable baffle types for New Zealand fishes.

4.5.7 Culvert embedment design

In addition to hydraulic considerations, an analysis of bed embedment should be implemented. Two general approaches are described in these guidelines for culvert embedment. The first approach is simple embedment and the second is stream simulation. Both approaches require design of appropriate embedment material, in order to remain in the culvert during design flood events. Properly designed culverts with simple embedment will provide many habitat benefits, as well as provide the maintenance of the sediment and bedload transport balance within natural streams. Stream simulation partially mimics the cross-sectional geometry of the stream within the culvert. The primary benefit of stream simulation is that fish passable velocities can be provided over a wider range of flows and a more widely varied aquatic habitat for native fish. The material used for the stream simulation within the culvert and immediately upstream and downstream of the culvert, must be designed to withstand higher velocities and shear forces than the adjacent portions of the stream. The two primary reasons for this are:

1. The culvert will have to convey flow that would ordinarily be conveyed by floodplains. This flow contraction results in higher velocities and greater depths during design flood flows.

2. Plants will not grow within the culvert. This eliminates the erosion protection afforded by aquatic and terrestrial plants within the stream and the floodplain. The additional roughness or flow resistance provided by the same plants cannot be provided within the culvert.

The approach for stream simulation within culverts is appropriate for very large culverts, bottomless culverts, and short-span bridges.

In the simple embedment approach, round culverts should have a minimum embedment that is ⅓ to ½ of the culvert diameter. Box culverts should have a minimum embedment height of 300 mm or 2 × D_{50}, whichever is greater. For bottomless culverts or culverts with full stream simulation, the critical shear stress method (see Section 4.5.8) should be applied.

Due to the scour patterns in bottomless culverts and stream simulation designs, embedment design will vary between the maximum applied shear stress for the channel bed and the equation developed for a rip-rap size that accounts for the local water velocity at the corner of the culvert entrance by Kerenyi and Pagan-Ortiz (2007):

$$D_{50} = \frac{K_r y_0}{(G_s - 1)} \left(\frac{{V_A}^2}{g y_0} \right)^{0.33} \tag{8}$$

Where:

D_{50} = rip-rap median size (50% finer) (m)

K_r = sizing coefficient equal to 0.38 from the best fit lab data, 0.68 for design curve that envelops the lab data

V_A = average velocity at the culvert entrance (m s^{-1})

y_0 = average flow depth at the culvert entrance before scour (m)

G_S = rip-rap specific gravity

g = acceleration of gravity (m s^{-2})

Figure 4-11 shows the scour pattern with no protection, while Figure 4-12 presents the use of rip-rap protection to reduce scour at culvert inlets.

Figure 4-11: Rectangular model with wing walls. Source: Kerenyi et al. (2003).

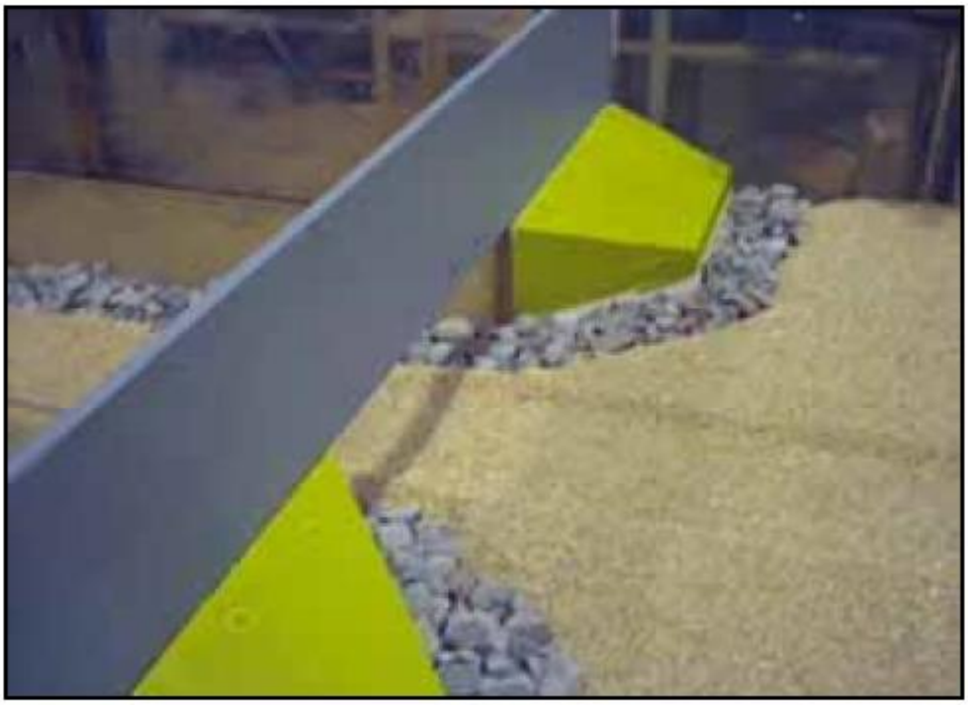


Figure 4-12: Rip-rap protection at a bottomless culvert with the MDSHA Standard Plan. Source: Kerenyi and Pagan-Ortiz (2007)

4.5.8 Culvert bed stability

The proposed culvert should maintain the dynamic equilibrium of sediment transport during 100-year ARI and 10-year ARI, which means sediment produced by the upstream reach should be transported through the proposed culvert. When the forces exerted by the flowing water on a particle exceed the resisting forces, the particle begins to move. The resisting forces include the weight of the submerged particle and the friction between particles.

There are a number of approaches to identify thresholds of the particles based on (a) critical shear stresses (Modified Shields Method), no cohesive materials (b) critical shear stresses, cohesive materials, or (c) the critical unit flow method (Kerenyi et al. 2003; Caltrans 2007; Hotchkiss and Frei 2007). The Modified Shields Method is typically applied in cases where slope is up to 5 percent, whereas for slopes ranging from 3 to 10 percent, the critical unit flow method is more appropriate.

To assess the stability of a channel bed, it is essential to evaluate the channel bed mobility in the upstream reach and estimate the maximum applied shear stress using HEC-RAS or a similar model, as well as determine the critical shear stress of the bed material. If the shear stress of the channel exceeds that of the bed material, it indicates potential instability. Conversely, if the shear stress of the channel is lower than that of the bed material, the bed can be considered stable.

If the channel bed is determined to be unstable, the proposed culvert should be designed to withstand a shear stress that is higher than the estimated maximum applied shear stress for the channel.

Critical shear stresses, no cohesive materials

Although there are multiple bed stability models designed for different types of bed material distribution and stream slopes, the Modified Shield's models are the preferred techniques. Critical shear stress values for a wide variety of sizes larger than 50 mm are established through research carried out in laboratory facilities and in the field, for slopes up to 10 percent.

The critical condition, that is the condition to be just less than that necessary to initiate sediment motion, is termed the threshold. It is assumed that the particle initiates motion (incipient motion) when the average shear stress slightly exceeds the critical shear stress. The average boundary shear stress exerted by a flowing body of water on its boundary is:

$$\tau = \gamma R S \tag{9}$$

Where:

τ	= average boundary shear stress (N m^{-2})
γ	= specific weight of water, 9810 (N m^{-3})
R	= hydraulics radius (m)*
S	= energy gradient (m/m)
*	R ≅ y, when the channel is wide or when flow within the culvert is not uniform

During the design flood peak, it is possible that the culvert may be flowing under pressure. This means that the culvert barrel is filled, and there is no free water surface for that section of the culvert. Since the assessment of bed stability requires the calculation of shear stresses, a method for computing shear stress under pressure conditions is necessary.

In theory, the critical shear stress for the bed material of the culvert remains the same, regardless of whether the culvert is flowing under pressure or not. The resistance to motion depends on the properties of the bed material. However, when pressure flow conditions exist, a modification is required to compute the applied shear stress because the culvert confines the water surface.

The applied shear stress under pressurised conditions is based on the hydraulic grade line instead of the free surface depth. The appropriate 'depth' for determining the applied shear stress under pressure flow conditions is the height of the energy grade line above the bed, subtracting the velocity head:

$$y = EGL - INV - \frac{V^2}{2g} \tag{10}$$

Where:

y	= depth (m)
EGL	= energy grade line elevation at point of analysis (m)
INV	= bed elevation at point of analysis (m)

V = velocity (m s^{-1})

g = acceleration of gravity (m s^{-2})

The critical shear stress for materials with a more uniform grading is determined using an equation that incorporates a characteristic grain size.

$$\tau_c = \tau_* \times (\gamma_s - \gamma) \times D_{50} \tag{11}$$

Where:

τ_c = critical shear stress (N m^{-2})

τ_* = dimensionless Shields parameter

γ_s = specific weight of sediment (N m^{-3})

γ = specific weight of water, 9810 (N m^{-3})

D_{50} = stone size for which 50%, by weight, of the bed is smaller (m)

Generally, a particular weight range of (24,500 to 25,900 N m^{-3}) is utilised for non-cohesive stone, but it is recommended to use a value that is specific to the site. Shield's parameter is expressed as a function of particle diameters as shown in Table 4-2. In the last column τ_c was calculated for different ranges of particle sizes (Federal Highway Administration 2010).

Table 4-2: Dimensionless Shield's parameter as a function of particle diameter.

Particle classification name	Range of particle diameters (mm)	τ_* (dimensionless)	τ_c (Pa)
Very Large Boulder	>2048	0.054	1790
Large Boulder	1024–2048	0.054	895
Medium Boulder	512–1024	0.054	447
Small Boulder	254–512	0.054	223
Coarse Cobble	127–254	0.054	111
Fine Cobble	63.5–127	0.052–0.054	53
Very Coarse Gravel	31.75–63.5	0.050–0.052	26
Coarse Gravel	16.0–31.75	0.047–0.050	12
Medium Gravel	7.87–16.0	0.044–0.047	5.7
Fine Gravel	4.06–7.87	0.042–0.044	2.71
Very Fine Gravel	2.00–4.06	0.039–0.042	1.26
Very Coarse Sand	0.99–2.00	0.029–0.039	0.47
Coarse Sand	0.48–0.99	0.033–0.029	0.27
Medium Sand	0.25–0.48	0.048–0.033	0.194
Fine Sand	0.12–0.25	0.072–0.048	0.145
Very Fine Sand	0.06–0.12	0.109–0.072	0.11

Natural bed materials are generally non-uniformly graded; therefore, the critical shear stress is determined by considering the interaction between larger and smaller particle sizes based on D_{84} and D_{50}.

$$\tau_C = \tau_* \times (\gamma_s - \gamma) \times {D_{84}}^{0.3} \times {D_{50}}^{0.7} \tag{12}$$

Where:

τ_C	= critical shear stress (N m^{-2})
τ_*	= dimensionless Shields parameter
γ_S	= specific weight of sediment (N m^{-3})
γ	= specific weight of water, 9810 (N m^{-3})
D_{50}	= stone size for which 50%, by weight, of the bed is smaller (m)
D_{84}	= stone size for which 84%, by weight, of the bed is smaller (m)

Critical shear stresses, cohesive materials

Critical shear stress on cohesive soils is primarily dependent on their cohesive strength and soil density, as cohesive soils are mostly composed of fine-grained materials. Cohesive strength is related to the plasticity index (PI), which is calculated by subtracting the plastic limit from the liquid limit of the soil. On the other hand, soil density is determined by the void ratio (e). The fundamental formula for determining critical shear stress on cohesive soils is as follows:

$$\tau_{c,soil} = (c_1 \times PI^2 + c_2 \times PI + c_3)(c_4 + c_5 \times e)^2 \times c_6 \tag{13}$$

Where,

$T_{c,soil}$	= critical shear stress (N m^{-2})
PI	= plasticity index
e	= void ratio
c_1, c_2, c_3, c_4, c_5, c_6	= coefficients from Table 4-3

Critical shear stress is expressed as a function of particle diameters as shown in Table 4-3. Soils can be classified as fine-grained, which includes GM, CL, SC, ML, SM, and MH, or coarse-grained, which is represented by GC. Clays (CH) are intermediate between these two groups.

The critical shear stress is increased by a higher soil unit weight, while a lower soil unit weight decreases critical shear stress. Table 4-4 is only applicable to soils that have a specific weight within 5% of the typical specific weight for a given soil class. For sands and gravels (SM, SC, GM, GC), the typical soil unit weight is around 1.6 ton m^{-3}, while for silts and lean clays (ML, CL), it is approximately 1.4 ton m^{-3}, and for fat clays (CH, MH) it is about 1.3 ton m^{-3}.

Table 4-3: Coefficients for critical cohesive soil shear stress. GM = Silty gravels, gravel-sand silt mixtures; GC = Clayey gravels, gravel-sand-clay mixtures; SM = Silty sands, sand-silt mixtures; SC = Clayey sands, sand-clay mixtures; ML = Inorganic silts, very fine sands, rock flour, silty or clayey fine sands; CL = Inorganic clays of low to medium plasticity, gravelly clays, sandy clays, silty clays, lean clays; MH = Inorganic silts, micaceous or diatomaceous fine sands or silts, elastic silts; CH = Inorganic clays of high plasticity, fat clays.

Unified Soil Classification	Applicable Range	C_1	C_2	C_3	C_4	C_5	C_6
GM	10≤ PI ≤20	1.07	14.3	47.7	1.42	-0.61	4.8×10^{-3}
	20 ≤ PI			0.076	1.42	-0.61	48
GC	10 ≤ PI ≤ 20	0.0477	2.86	42.9	1.42	-0.61	4.8×10^{-2}
	20 ≤ PI			0.119	1.42	-0.61	48
SM	10 ≤ PI ≤ 20	1.07	7.15	11.9	1.42	-0.61	4.8×10^{-3}
	20 ≤ PI			0.058	1.42	-0.61	48
SC	10 ≤ PI ≤ 20	1.07	14.3	47.7	1.42	-0.61	4.8×10^{-3}
	20 ≤ PI			0.076	1.42	-0.61	48
ML	10 ≤ PI ≤ 20	1.07	7.15	11.9	1.48	-0.57	4.8×10^{-3}
	20 ≤ PI			0.058	1.48	-0.57	48
CL	10 ≤ PI ≤ 20	1.07	14.3	47.7	1.48	-0.57	4.8×10^{-3}
	20 ≤ PI			0.076	1.48	-0.57	48
MH	10 ≤ PI ≤ 20	0.0477	1.43	10.7	1.38	-0.373	4.8×10^{-2}
	20 ≤ PI			0.058	1.38	-0.373	48
CH	20 ≤ PI			0.097	1.38	-0.373	48

Table 4-4: Sub-soil type and corresponding critical shear stress values.

	Cohesive				
Sub-soil type	Fine grained		Clay	Coarse grained	
PI range	20 < PI	10 < PI < 20	20 < PI	20 < PI	10 < PI < 20
Stress range (N m^{-2})	3.9–4.5	1.3–4.5	5.7	7.1	4.6–7.1

It should be noted that the critical shear stress required for cohesive sediment deposition is typically lower, sometimes significantly lower, than the shear stress needed for cohesive sediment erosion. This is particularly possible when there is enough time between events for the fine sediment to consolidate.

Critical unit flow

The critical unit flow method can be employed when the slope ranges from 3 to 10%. This approach serves as an alternative measure of bed of stability when it is challenging to measure or define the depth due to high relative size of roughness elements to the depth. It involves calculating the unit flow and critical unit flow. Unit flow is defined as the flow above the active channel bed divided by the width of the active channel bed:

$$q = \frac{Q_a}{w_a} \tag{14}$$

Where:

q = unit flow ($m^3 s^{-1}/m$)

Q_a = active channel flow ($m^3 s^{-1}$)

w_a = active channel bed width (m)

In a culvert, the differentiation between active channel and other channel components, such as a floodplain, is not practical. As a result, the active channel flow represents the total flow, and the active channel width corresponds to the flow top width.

In the case of materials that are uniform, the critical flow is determined by calculating it with reference to the characteristic grain size:

$$q_{c-D_{50}} = \frac{0.15 g^{0.5} {D_{50}}^{1.5}}{S^{1.12}} \tag{15}$$

Where:

$q_{c-D_{50}}$ = critical unit flow to the D_{50} particle size ($m^3 s^{-1}/m$)

D_{50} = median or 50th percentile particle size (mm)

g = acceleration of gravity ($m\ s^{-2}$)

S = bed slope ($m\ m^{-1}$)

To adapt to the more typically nonuniform bed materials found in a natural channel, the critical unit flow for entraining the D_{84} particle size is determined by:

$$q_{c-D_{84}} = q_{c-D_{50}} \left(\frac{D_{84}}{D_{50}}\right)^{b} \tag{16}$$

Where:

$q_{c-D_{84}}$ = critical unit flow to the D_{84} particle size ($m^3 s^{-1}/m$)

D_{84} = 84th percentile particle size (mm)

D_{50} = median or 50th percentile particle size (mm)

The exponent 'b' represents a parameter that measures the extent of particle size distribution within the channel bed. It quantifies the impact of smaller particles that are hidden and larger particles that are exposed to the flow on particle entrainment. The formula to compute the exponent is as follows:

$$b = 1.5 \left(\frac{D_{16}}{D_{84}}\right) \tag{17}$$

Where:

D_{84} = 84th percentile particle size (mm)

D_{16} = 16^{th} percentile particle size (mm)

The equations to calculate $q_{c-D_{84}}$ and b are limited to the conditions in Table 4-5.

Table 4-5: Parameter ranges for critical unit flow for D_{84}.

Parameter	Low	High
Slope (%)	3.6	5.2
Width (m)	6.1	11
D_{16} (mm)	32	58
D_{50} (mm)	72	140
D_{84} (mm)	156	250

If the unit flow is less than or equal to the critical unit flow, the designer proceeds to check the culvert embedment stability at Q_p. If not, check channel bed mobility at high passage flow.

4.5.9 Culvert aprons

Aprons are designed to prevent scour/erosion, manage hydraulic jumps, facilitate non-erosive flow into the downstream channel, and ensure fish passage. The design of aprons is influenced by factors such as culvert size, shape, longitudinal grade, stream characteristics upstream and downstream of the culvert, and the fish species that need to pass through, particularly for upstream migration. For guidance and design methods related to culvert aprons, the following references provide relevant information:

- HEC-11 SI, Design of Rip-rap Revetment, US FHWA (Brown and Clyde 1989).
- HEC-14, Hydraulic Design of Energy Dissipators for Culverts and Channels, US FHWA (Thompson and Kilgore 2006).
- HEC-15 SI, Design of Roadside Channels with Flexible Lining, US FHWA (Kilgore and Cotton 2005).
- HEC-23 Bridge Scour and Stream Instability Countermeasures Volumes 1 and 2, US FHWA (Lagasse et al. 2009).
- HDS-6, River Engineering for Highway Encroachments, Highways in the River Environment, US FHWA (Richardson et al. 2001).
- NCHRP-568, Rip-rap Design Criteria, Recommended Specifications and Quality Control (Lagasse et al. 2006).
- USDA Natural Resources Conservation Service, Part 654 National Engineering Handbook, Stream Restoration Design, Technical Supplement 14C,(Natural Resources Conservation Service 2007).
- Mile High Flood District, Urban Storm Drainage Criteria Manual Volume 1 (Urban Drainage Flood Control District 2016a) and Volume 2 (Urban Drainage Flood Control District 2016b).

- Mile High Flood District, Specifications, Division 31 Rip-rap Boulders, and Bedding (Urban Drainage Flood Control District 2017).
- Open Channel Hydraulics (Chow 1959).

There are two general approaches to culvert apron design:

- the application of empirically-developed apron configurations, and
- bespoke design using the accepted Standard Step method.

One challenge with empirically-developed culvert aprons is that they typically do not consider fish passage, and the physical models used in the studies were not tested at flows suitable for fish movement. As a result, the empirically-developed culvert apron configurations need to be assessed for suitability and may require adjustments. **It is important to note that empirically-developed culvert apron designs that rely solely on rock mass and roughness to control energy losses upstream or downstream of culverts are inadequate when considering fish passage requirements**.

Apron design for fish passage

In addition to providing scour protection, the aprons for large culverts need to create transition zones that reduce the abruptness of the hydraulic geometric transition in the channel section, approaching and exiting the culvert. Contraction of flow at the upstream end of the culvert can result in higher water velocities at the very edges of the culvert, potentially preventing fish from exiting the culvert. Rapid expansion of the flow and drop in water surface level at the downstream end of the culvert also causes locally increased water velocities that can prevent native fish from entering the culvert. By carefully designing the culvert aprons, the culvert inlet contraction and outlet expansion issues can be reduced. Interrupting the bedload and sediment transport balance at a crossing can result in the following:

- Increased maintenance requirements due to sediment accumulation and/or erosion around the culvert.
- Initiate or exacerbate channel erosion and instability downstream of the culvert.
- Flooding upstream of the culvert, due to sediment accumulation.
- Deposition within the culvert, due to bedload and sediment transport imbalance, which can create barriers to fish passage.
- Downstream erosion, due to interruption of bedload and sediment transport, which can cause the formation of barriers to fish passage.

Figure 4-13 shows a cross-section of an apron for a large triple box culvert, intended to provide scour protection for flows up to the 100-year climate adjusted event, and accommodate fish passage through the middle box at ½ of the 2-year ARI flow.

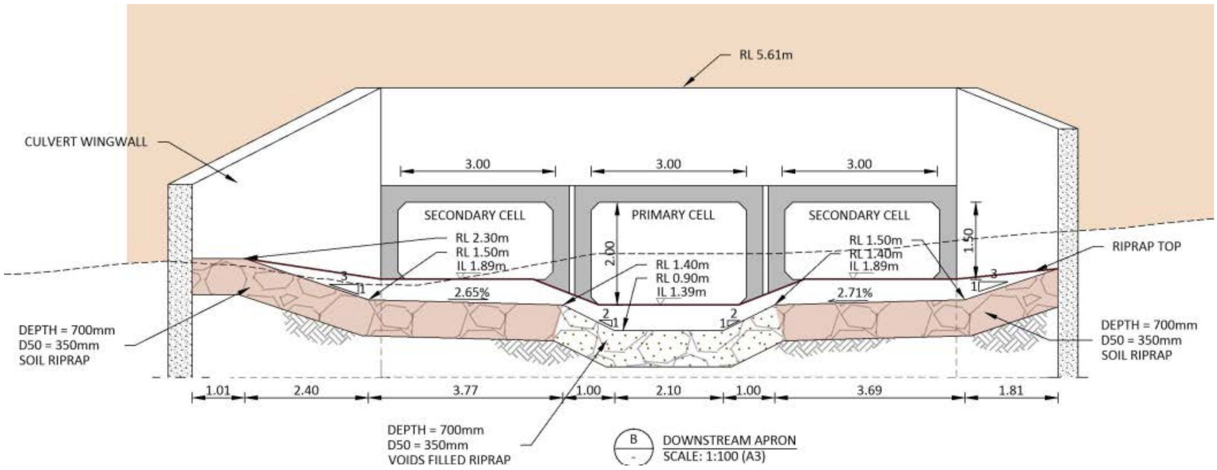


Figure 4-13: Cross-section of scour-protected apron for large triple box culvert. Designed for 100-Year climate-adjusted flows and fish passage at ½ of 2-year ARI flow.

The apron is depressed to form a stilling basin. Note the use of soil rip-rap and voids-filled rip-rap to prevent low flow water surface levels from being below the top of the rip-rap and support plant growth, which will be discussed later in this section. Figure 4-14 shows the plan view of the upstream apron. In this example, a stream and a steep tributary form a confluence within the culvert apron. The stream and its tributary were highly modified to form part of an agricultural drain network.

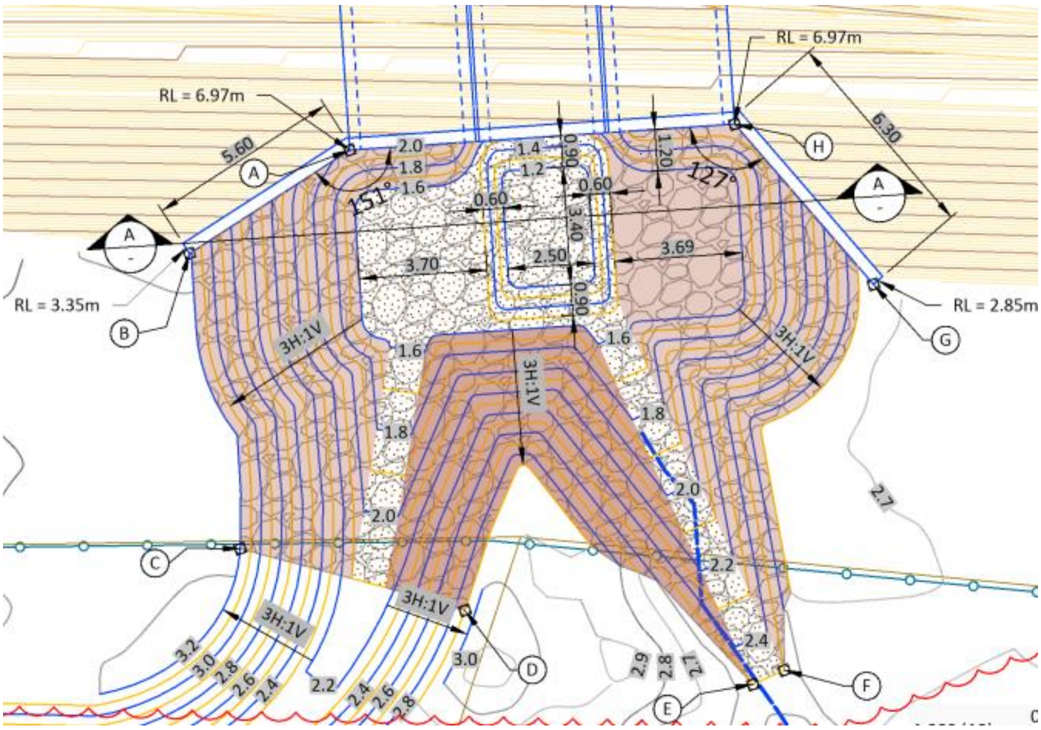


Figure 4-14: Plan view of upstream apron with stream and steep tributary confluence.

Although the culvert and aprons were designed using 1D modelling, the depth-averaged water velocities within the aprons were verified using the associated 2D flood model, which employed a terrain mesh with variable cell sizes. The cell size employed within the aprons was significantly smaller in comparison to the cell size used in the floodplain areas upstream and downstream of the culvert and aprons.

The appropriate cell size necessary to adequately represent flow patterns within the aprons will depend on the specific 2D modelling software being utilised. Figure 4-15 shows the depth averaged water velocities from the HEC-RAS 2D model within the upstream culvert apron during the climate change ½ of the 2-year ARI flow. Note how the depth averaged water velocities decrease as the flow passes through the upstream stilling basin and the lower centre box has inflow depth-averaged water velocities less than 0.3 m s^{-1} on either side of the inlet. While stilling basins can be applied very effectively at the upstream end of a culvert, caution is required to avoid increasing the hydraulic grade too much in the culvert, due to the rise in the water level that can occur in the apron due to specific energy. Generally, specific energy is not an issue with the fish passage design flow.

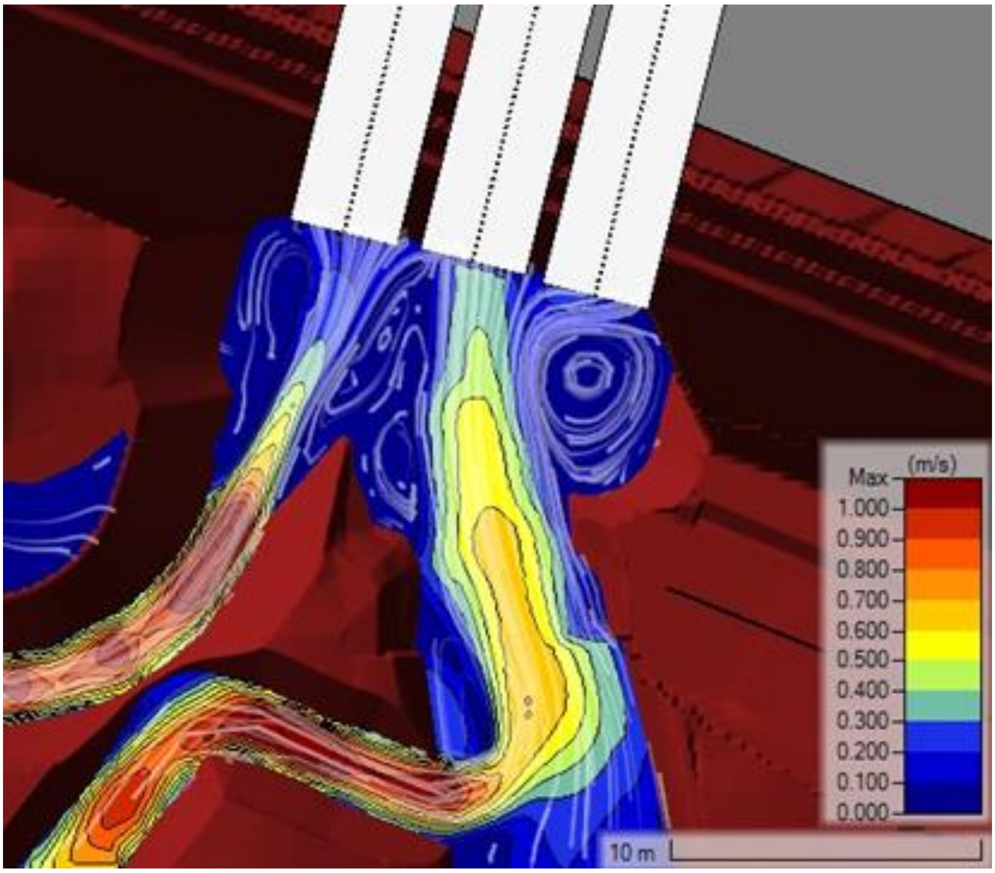


Figure 4-15: HEC-RAS 2D depth-averaged water velocities in the upstream culvert apron at the climate-adjusted ½ of 2-year ARI flow.

Figure 4-16 shows the vertical distribution of water velocities, with the depth-average velocity marked in red. As indicated in the figure, the lower 40% of the depth will have water velocities lower than the depth-averaged water velocity. Adding a stilling basin increases the depth of low water velocity, creating resting habitat for fish.

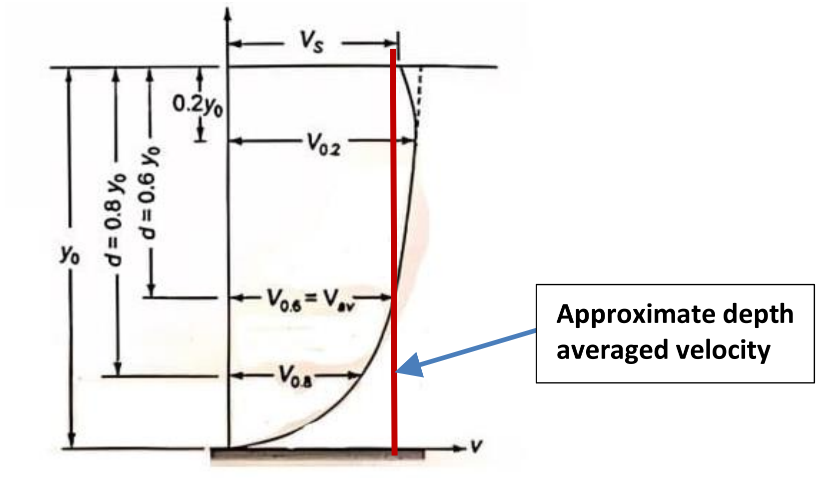


Figure 4-16: Vertical distribution of water velocities with depth-average velocity highlighted (red). Demonstrates lower water velocities in the lower 40% of depth.

The benefits of the stilling basins are to reduce water velocities at the culvert inlet and create a low water velocity zone upstream of the culvert inlet to receive the fish once they exit the culvert. Sediment and bedload deposition can be expected in the stilling basins, reaching equilibrium over time.

Figure 4-17 shows the downstream apron for the same culvert. Again, the apron is shaped to form a stilling basin. An outlet from a large stormwater treatment and attenuation wetland was incorporated into the culvert apron, in this case. The use of a stilling basin at the culvert outlet provides increased water depth and a slight increase in water surface level due to specific energy, just as it did in the upstream apron. The increased water level at the culvert outlet due to specific energy helps to maintain a lower hydraulic grade through the culvert and helps maintain the outlet control of the culvert to maximise potential for upstream fish movement.

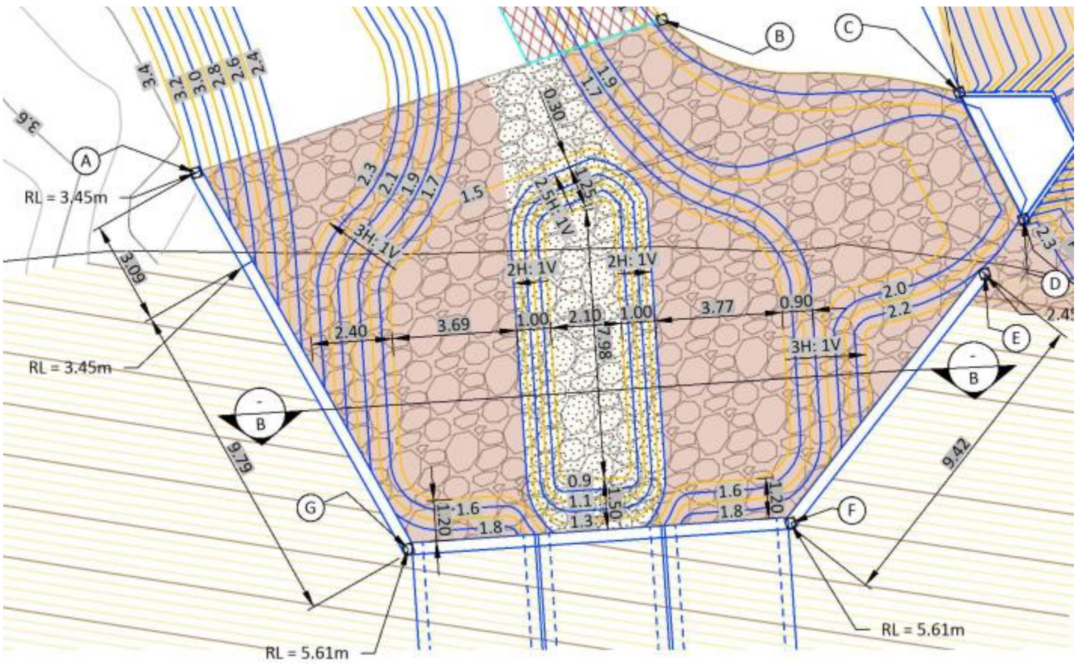


Figure 4-17: Plan view of downstream apron with stream and steep tributary confluence.

The effect on water velocities of the downstream apron being configured as a stilling basin can be seen in Figure 4-18, which shows the 2D model output at the fish passage design flow. Note the very low water velocities through the apron, extending to the culvert outlet.

As with the upstream apron, bedload end sediment deposition is expected to accumulate until equilibrium is established. These bedload and sediment deposits will support wetland vegetation in the shallower areas, outside of the primary channel.

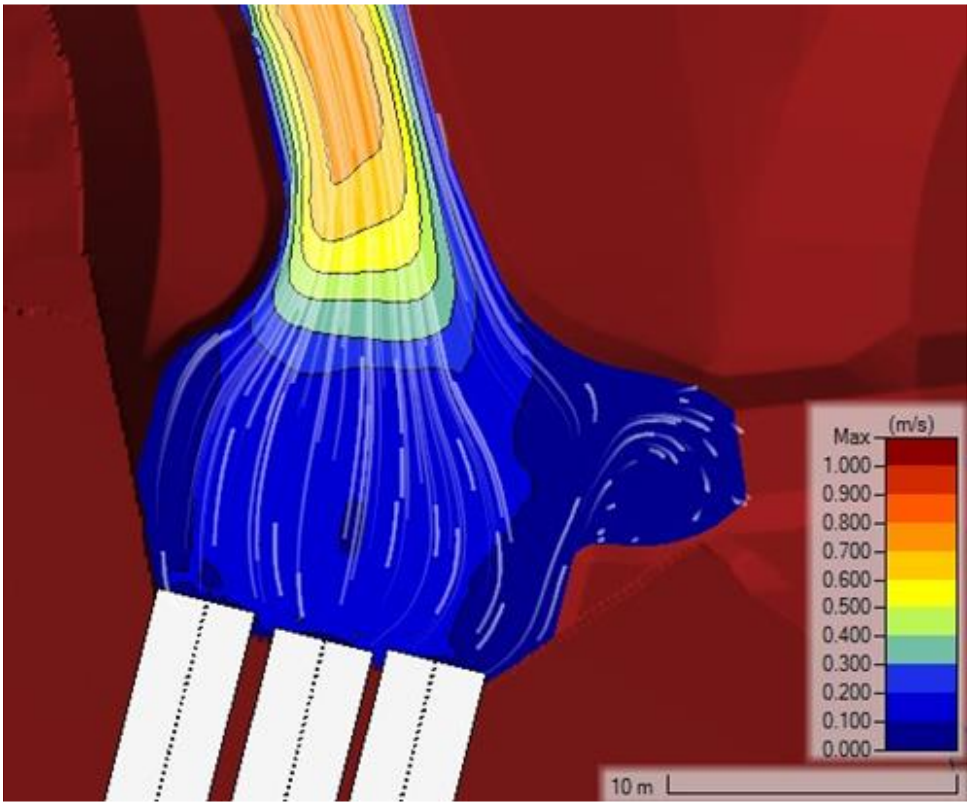


Figure 4-18: Depth-averaged water velocities in a downstream culvert apron at the climate-adjusted ½ of 2-year ARI flow.

General geometric requirements for culvert aprons

The following are general requirements for culvert aprons, where fish passage must be accommodated:

- For culverts with an internal cross-sectional area of 3.4 m^2 (lower limit of Waka Kotahi NZTA Bridge Manual culvert coverage) or more, it must be demonstrated that the water velocities at the inlets and outlets have been factored into the design of a fish passable culvert.
- For culverts with an internal cross-sectional area smaller than 3.4 m^2 (combined for multicell) where fish passage must be provided, the downstream apron must include an appropriately sized stilling basin.
- Culvert aprons must not result in an increase in hydraulic grade through the culvert when conveying the fish passage design flow.

- Culvert aprons must not exacerbate increased water velocities at the inlet and outlet of the culvert.

Rip-rap for fish passage applications

While rip-rap provides an excellent flexible scour countermeasure when correctly applied, low to normal flows in smaller perennial streams may flow through it, more than over it. Rip-rap is approximately 30% interstitial void space by volume and can cause a significant barrier to fish passage, when all or most of the flow is moving through the interstitial void in the rip-rap. To facilitate fish passage in locations where rip-rap is required, allow planting, and create a closer simulation of a natural stream bed, soil rip-rap or voids-filled rip-rap must be used. Soil rip-rap is a mixture of 65% rip-rap with 35% in-situ soils (by volume), mixed at the work location and then placed. Where it is necessary to avoid the introduction of fine silt and/or clay-sized sediment into a stream, voids-filled rip-rap can be applied. Voids-filled rip-rap is virtually the same as soil rip-rap, except sand, gravel, and small crushed rock or cobbles are substituted for the in-situ soil. Generally, the largest particle size in the voids filling material should not exceed 20% of the D_{50} of the rip-rap that it is being combined with. When placing rip-rap structures in sand or gravel bed streams or rivers, it is appropriate to use the in-situ bed material as the void filler. By filling the interstitial voids in the rip-rap, the need for rip-rap bedding or geotextile under the rip-rap is eliminated. Soil rip-rap has been successfully applied in the US, Australia, and New Zealand. Appendix G provides examples of soil rip-rap applications for aprons and stream works, while Appendix H outlines specifications for scour protection.

4.5.10 Worked examples

Four worked examples demonstrating the workflows and methodologies described above for standard culvert designs have been prepared and are presented in Appendix C to Appendix F.

4.5.11 Culvert design: Steep stream crossings

For the purposes of these guidelines, a stream is considered steep if:

- the **average** longitudinal grade of the stream is greater than 0.4% over 500 m, extending 250 m upstream and downstream of the culvert location,
- there are sections of the stream reach that exceed a 1.0% longitudinal grade (e.g., riffles, in pool-riffle systems),
- the streambed consists of gravel, cobbles, or boulder substrate, and
- the stream reach conveys supercritical flow through all or part of the reach extending 500 m upstream and downstream of the proposed culvert location (modelled Froude numbers exceeding 0.8 should be considered as equivalent to being supercritical for the purpose of defining a steep stream crossing in these Guidelines).

The methodology for designing a fish passable culvert in a steep stream is the same as it is for designing a culvert in a stream that exceeds the limitations of the Basic Methodology (Section 4.5.2). The primary potential differences are the target species used to define the fish passage water velocity criteria and requirements for determining culvert embedment.

Fish passage criteria: water velocity

As stream gradient and distance from the coast increase, fish community composition will change. As such, while īnanga are an appropriate target species for defining the fish passage criteria for low gradient streams (Section 4.5.6), they may not be present in some steeper streams. A suitably qualified freshwater ecologist should determine whether īnanga are expected to be naturally present at a site. Where īnanga are expected to be naturally present, the same fish passage water velocity criteria will apply as for the standard methodology (Section 4.5.6). Where īnanga are not expected to be naturally present, an alternative benchmark species for the fish passage water velocity criteria should be determined by a suitably qualified freshwater ecologist.

Species commonly encountered in steeper streams at higher elevations and distance from the coast include longfin and shortfin eels, banded kōkopu, and kōaro. Non-diadromous bully species, e.g., Cran's bully or upland bully, may also be present. There are currently limited swimming endurance data available for these species necessary for determining fish passage water velocity criteria following the standard methodology set out in Section 4.5.6. However, Crawford et al. (In review) compared critical swimming speeds of native fishes in New Zealand and showed that longfin elvers, banded kōkopu, and two bully species (common bully and redfin bully) have similar critical swimming speeds to īnanga. This indicates that even in steep streams where īnanga are not present, based on current best available information, īnanga may still be an appropriate benchmark species for determining fish passage water velocity criteria for culvert design in steep streams. Where a suitably qualified freshwater ecologist determines that īnanga are not an appropriate benchmark species, data will be required to determine the swimming endurance of the selected target species. Swimming endurance data for longfin elvers are available and may offer a suitable benchmark (Figure 4-19). Generally, the water velocity distribution within the culvert needs to be such that fish can find passable areas within the culvert as they do in the stream. Due to the smooth culvert sides above the embedment, and the more regular bed form within the culvert, this will generally require a slower average water velocity in the culvert than the adjacent stream to achieve the critical lower portion of the water velocity distribution in the stream.

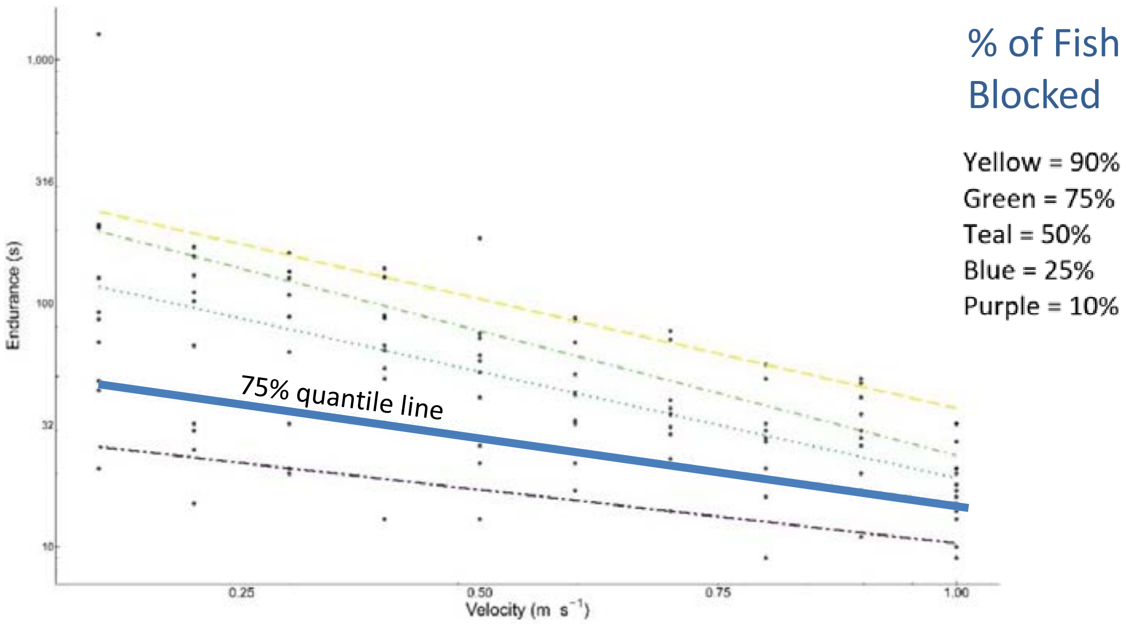


Figure 4-19: Swimming speeds and endurance for longfin elvers. Quantile regression was used to establish relationships between swimming speed and endurance for different percentages of individuals.

Culvert embedment design

While embedment material within the culvert is designed using the same methodology as the standard method for larger streams (Section 4.5.7), the potential for a large size requirement for rip-rap requires some consideration. Culvert embedment for round culverts remains a minimum of ⅓ of the diameter to a maximum of ½ of the culvert diameter. The chord width of a round culvert at the streambed level must be at least as wide as the width of the streambed. The embedment depth of box culverts remains 2 × D_{50} of the properly sized embedment material, with a minimum embedment depth of 300 mm. If the calculated rip-rap size (D_{50}) exceeds 20% of the height of a round culvert or box culvert, then the culvert will either need to be:

1. Enlarged to reasonably accommodate the larger material, or
2. The culvert will be embedded, without placing embedment material within the culvert, allowing the embedded depth to be filled through the bedload and sediment transport process.

In case 2, there are limitations on how deep an open culvert can be embedded within the stream, without destabilising the stream. To avoid bed and bank destabilisation, open embedment of round culverts is limited to a maximum culvert diameter of 1.2 m. Box culverts can be embedded without placing embedment, to a depth of the average size of the visible bed material within the steeper portions of the stream. In cases where the **average** size of the visible bed material is greater than 450 mm, it is recommended that a bridge is used for the channel crossing.

4.6 Fords

4.6.1 Overview

Fords must not impede the passage of fish without approval from DOC (FFR83 s42(1)). Consequently, it is essential that all new and replacement fords are designed to incorporate and provide for fish passage on an ongoing basis from the outset of the design process. Fords can be a very problematic stream crossing for fish passage, as they often combine many of the negative features of culverts and weirs, involve modification of the stream bed, and allow vehicle access to the stream. Consequently, wherever possible, the construction of new fords should be avoided, and alternative river crossings used.

Fords that do not have a raised roadway typically still involve modifications to the river bed that reduce substrate and hydraulic complexity, and increase water velocities over the ford (e.g., Figure 4-20). Increasingly, fords have been raised above the natural stream bed to help mitigate disturbance of the stream bed during vehicle crossings. These crossings are also sometimes termed causeways. Simple raised roadway fords (e.g., Figure 4-21) often impede fish passage by a combination of a steep downstream face, a sharp crest, shallow water, and high water velocities over the ford. They also impact on geomorphic processes, disrupting sediment transport. These impacts can be reduced by incorporating culverts into the ford to pass the stream under the roadway under low to moderate flow conditions (e.g., Figure 4-22). In this case the water velocity or depth in the culvert, or the length of the culvert, may still impede fish passage if poorly designed.

Figure 4-20: Low profile ford crossing.

Figure 4-21: Raised roadway ford crossing. The vertical drop on the downstream side will block fish movements. Shallow water depth and elevated water velocities across the ford pavement can also impede movements.

Figure 4-22: Raised roadway ford crossing with culverts. At low flows, water passes through the culverts in the ford, rather than over the roadway. The small culverts used in this ford severely constrict the river channel and will result in accelerated water velocities through the culvert barrel that may be impassable to fish.

4.6.2 Design principles

Best practice is to avoid the use of fords for stream crossings as they are the least preferred crossing type from a fish passage perspective and do not prevent vehicles or animals from entering the waterway. Where a ford is deemed necessary, the principles of good fish passage design (Section 4.1) should be applied. Low profile and standard raised roadway ford designs should be avoided. Causeway ford designs incorporating culverts are the minimum standard for fords. The objective is to ensure that a continuous pathway for fish passage is maintained across the structure over the fish passage design flow range.

4.6.3 Initial assessment and design flows

A priority for the initial site assessment is to ensure that an alternative stream crossing type cannot be used. Where a ford is deemed necessary, the standard catchment-scale review and site reconnaissance process should be undertaken to evaluate channel stability and the operating range for the structure.

The migration periods of the target fish species identified in the initial assessment should be used to inform the fish passage design flow range (see Section 3 on setting objectives and performance standards). This sets the range of flows over which the hydraulic performance standards for the structure must be met. The fish passage low flow (Q_L) is the lowest flow at which fish passage must be provided and as a rule of thumb can be set at the 95% exceedance flow. The fish passage high flow (Q_H) is the highest flow at which the hydraulic performance standards for fish passage should be met. A rule of thumb is to use the 50% of the 2-year ARI flow for Q_H but this should be defined on a site-specific basis determined by the biological requirements.

The design peak flood flow (Q_P) is a reasonable estimation of the highest flow that the ford should be designed to pass without causing a significant increase in upstream flooding. The appropriate standard for Q_P should be determined with reference to relevant regional plan rules, local drainage standards and technical design guidelines for roadways and infrastructure. For fords there will also likely be thresholds set for the return interval flow event (Q_i) that will inundate the road. Using this knowledge, objectives and performance standards should be set (Section 3).

4.6.4 Ford design

Ford design should follow the guidelines for culvert design in Section 4.5. The following key features must be incorporated in the design:

- Reduction of the channel cross-sectional area at the ford over the fish passage design flow range should be avoided or minimised.
- Where stream size dictates (i.e., bank-full width is too great for a single span culvert), multiple box culverts may be required to span the full wetted width of the stream without significantly constricting cross-sectional area.
- Circular culverts should be avoided where multiple barrels are required.
- Substrate must be maintained through the full length of the culverts and remain stable across the fish passage design flow range.
- Alteration of natural stream channel alignment should be avoided or minimised.
- Alteration of natural stream channel gradient should be avoided or minimised.

- Determine the design water velocities over the fish passage design flows (Q_L to Q_H) to facilitate passage of the target fish species and sizes. Determine the water velocity requirements for the smallest sized fish of each species to require passage, and then choose the lowest of these velocities for design. Alternatively use the adjacent stream as a reference for defining water velocity requirements.
- Check the water depths associated with the design velocities and ensure that they are deep enough to allow passage of the target fish species and life stages. Where possible provide heterogeneity of water depth through all elements of the structure.
- Check that the slope of each part of the culvert, and the transition between slopes, does not provide an impediment to passage.
- Ensure that the surface of the ford is roughened (e.g., through embedding rocks) to facilitate passage of fish over the ford when flows overtop the structure.
- The lateral profile of the ford should be V-shaped to ensure that wetted margins are maintained across the ford when it is overtopped during elevated flows.
- Drift deck designs may be suitable where they are installed in a way that is consistent with the design guidance above.
- Where fords are used as a temporary crossing, they must be installed in a way that is consistent with the design guidance above to provide unimpeded fish passage and removed within 2 months of installation.

Final design & construction

Once structural design is completed, construction drawings and specifications can be prepared and finalised for contracting. Subsequently, construction of the stream crossing occurs. Specialists involved in the design should continue to be informed of the construction process and be involved as necessary to help negotiate any challenges as they arise on site. It is important to ensure that the project is built to specification and that any departures from that specification are agreed with the design team. All relevant consents required for dewatering and construction should be in place, along with appropriate plans for sediment and pollution control during the construction phase. These rules will generally be determined in regional plans under the requirements of the RMA.

4.7 Weirs

4.7.1 Overview

Weirs are inherently an interruption to the slope of the stream bed. Weirs may combine several obstacles to upstream and downstream passage of fish including:

- fall heights that prevent swimming species from migrating upstream;
- crest shapes that may be insurmountable to climbing species;
- shallow water depths either upstream or downstream of the weir;
- increased water velocities, and;
- inappropriate attraction flows.

Furthermore, the backwater effect upstream of weirs inundates and alters instream physical habitat, typically resulting in a shift towards slower flowing and deeper habitats. Consequently, where possible, the installation of new weirs should be avoided.

Weirs may be built for a range of purposes, including flow gauging, flood control, and maintenance of a prescribed upstream water level (e.g., for abstraction). Flow gauging weirs have relatively strict technical requirements for maintaining the accuracy of hydrological measurements, imposing limitations on the shape of the weir and any possible fish passage provisions that can be included. However, recognition of the environmental impact of gauging weirs, in combination with technological improvements in other gauging techniques means that this type of weir is increasingly redundant. Consequently, installation of new flow gauging weirs should largely be unnecessary. Where maintenance of a minimum upstream water level is the intended purpose of the weir (i.e., a head control structure), a wider variety of options for providing fish passage can be considered. Some of the key features of fish friendly weir design are discussed below.

Relatively little work has been undertaken in New Zealand to specifically evaluate weir design requirements for passage of native fish species. Consequently, recommendations in these guidelines are based on international good practice in combination with local experience and expert interpretation of experimental work that has been carried out on fish ramp designs in New Zealand.

4.7.2 Design principles

Conventional weir designs that incorporate smooth concrete bottoms and steep hydraulic drops are unsuitable for providing fish passage and should be avoided where practicable. Good practice where the objective of a weir is simply to maintain a minimum headwater level is to use a full width rock ramp fishway as an alternative to a traditional weir structure (see Section 5.5.3). A rock ramp can be used to disperse the hydraulic head over a greater distance than a vertical or very steeply inclined concrete weir by keeping the hydraulic gradient gentle (e.g., 1:15 to 1:30). Low-gradient rock ramps exhibit a high level of structural diversity, imitating natural stream conditions, and provide a multitude of opportunities for passage of different organisms.

Where more nature-like solutions are not practicable, there are several design principles that should be considered:

- Vertical and steep hydraulic drops should be avoided.
- Undershot weirs should be avoided.
- Broad-crested weir designs should be used.
- Weir crests should be rounded.
- The weir should have a V-shaped lateral profile providing shallow, low velocity wetted margins on the weir face across the fish passage design flow range.
- The slope of the downstream weir face should be minimised.
- The use of smooth concrete on the weir face should be avoided or minimised.
- Vertical wing walls should be avoided.
- Back watering of upstream habitats should be minimised.

The NES-F sets out minimum design standards for weirs to meet permitted activity status (see Section 2.4.3) that should be used as a benchmark for weirs where a rock ramp fishway cannot be used. Weirs must also meet the requirements of the FFR83 (Section 2.1).

4.7.3 Initial assessment and design flows

The standard catchment-scale review and site reconnaissance process should be undertaken to evaluate channel stability and the operating range for the structure. The initial assessment of the site and purpose of the structure should consider whether a weir is the most suitable option, or whether the desired outcome can be achieved by some other means with a lower impact on river connectivity. For example, can the desired purpose be achieved by pumping from the stream to off-stream water storage, or deriving a flow rating curve at a morphologically stable site that does not require the construction of a weir?

Where a weir is determined to be the only practicable solution, the suitability of a full width rock ramp fishway for achieving the required headwater level should be evaluated. Only where this is not practicable should a more conventional weir design be selected.

The objectives and performance standards for the site should be identified (see Section 3) including the species and life stages of interest, and the smallest size fish of each species that require passage. Fish passage design flows should be determined for the target species.

4.7.4 Weir design

Weir Type

Where practicable the weir should be built as a rock ramp fishway (e.g., Figure 4-23; Figure 5-9). Details on the design of rock ramp fishways suitable for New Zealand fish species are provided in Section 5.5.3. Full river width rock ramp fishways are the optimal design for overcoming low-head barriers (≤1 m) on many river types and are also suitable in many locations for larger head differences (<4 m) where sufficient stream length is available to accommodate the low slope designs (Figure 4-24). Where a more conventional weir is required, broad-crested weir designs with a sloped downstream face should be chosen. Guidelines on key design features of these weirs is provided in the following sections. Incorporation of partial width rock ramp fishways (see Section 5.5.3) or bypass structures (see Section 7.3.5) should also be considered as an integral component of conventional weir designs.

Undershot weirs (sluice gates) should be avoided as they have been shown to subject fish to considerably higher pressures, shear stresses, and risk of physical strike, and have been found to be significantly more problematic for fish to negotiate than overshot weirs (Baumgartner et al. 2006). Australian studies have shown that downstream-drifting larvae of Murray cod (*Maccullochella peelii*) and golden perch (*Macquaria ambigua*) have a significantly higher mortality associated with passage through an undershot low-head weir than an overshot low-head weir (Baumgartner et al. 2006). Downstream movement of the larval stage is common among New Zealand's native fishes, and it is reasonable to assume that similar outcomes would occur at undershot weirs here. Furthermore, several of our upstream migrating native species can climb wetted surfaces (McDowall 2000). Undershot weirs will prevent the use of this movement strategy, but passage at overshot weirs may be achievable where the right conditions are provided. **Where possible, therefore, overshot weir designs should be chosen** (Harris et al. 2017).

Figure 4-23: A rock ramp style weir on the Waipa River at Otorohanga that also has a fish pass along the true left bank. The fish pass provides passage at low flows when the large rocks forming the downstream face of the weir are exposed and swimming species cannot surmount the weir. Photo credit: Eleanor Gee.

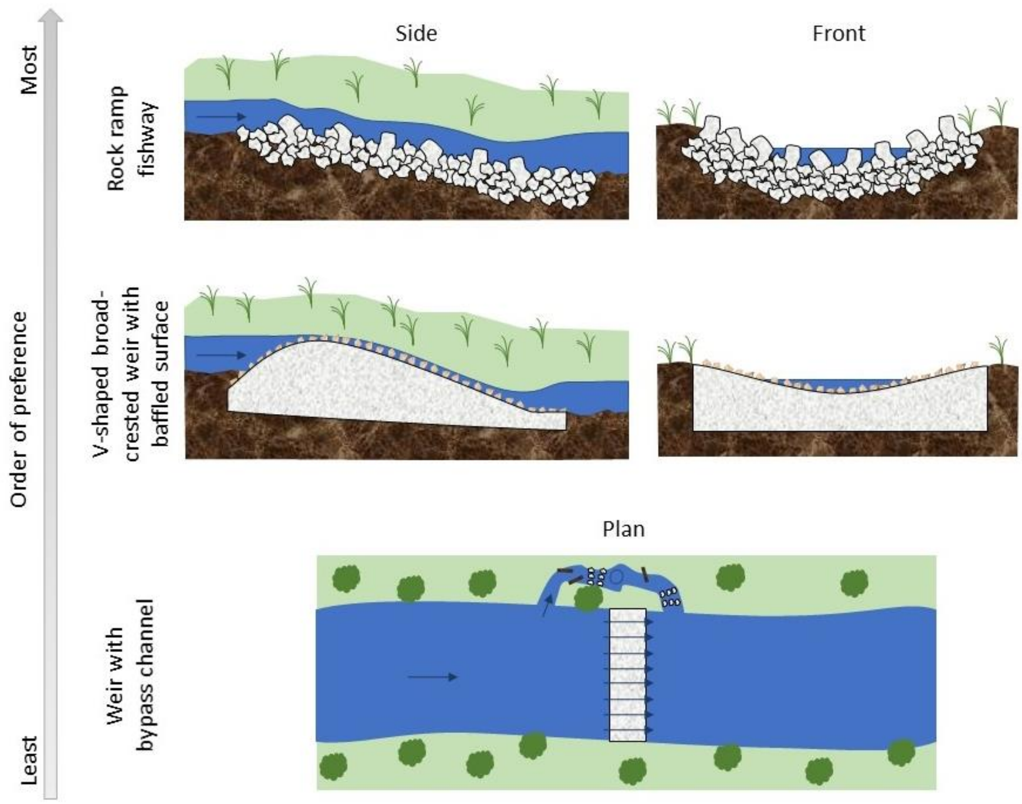


Figure 4-24: Order of preference for head control structure designs, based on the degree of connectivity each design facilitates.

Lateral profile

Rock ramp and conventional weir designs should have a V-shaped lateral profile that rises towards the river banks producing zones of calmer flow in the marginal areas and a low-flow channel towards the centre of the weir (Figure 4-25). The angle of the V-shape should generally be in the range of 5-10° for a full width weir (Figure 4-25).

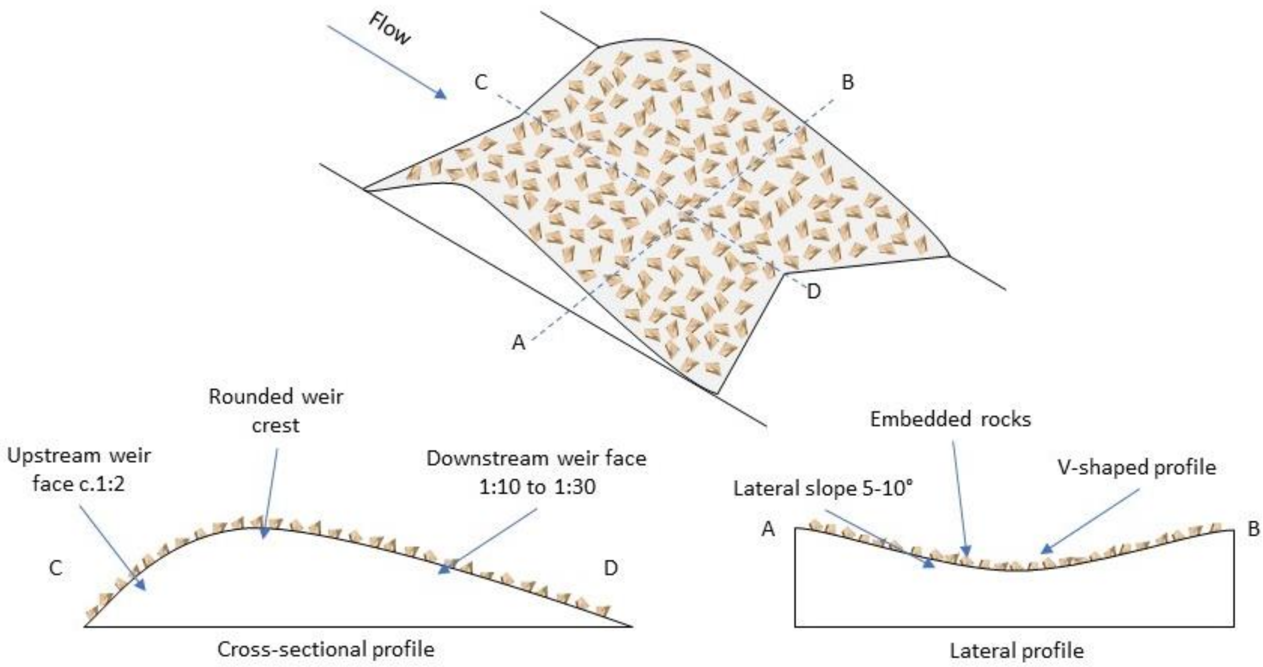


Figure 4-25: Key features of a conventional weir design for fish passage.

Downstream weir face

The gradient of the downstream weir face should be gentle. The slope of rock ramp fishways should generally be between 1:15 and 1:30, with the 1:30 design recommended for weaker swimming native fish species such as īnanga (see Section 5.5.3 for further details on determining slope of rock ramp fishways). The slope of conventional weir designs should be minimised and generally be less than 1:10 for fall heights of ≤1 m or less than 1:15 for fall heights of 1–4 m (Figure 4-25).

The design velocity on the downstream face of the weir should provide for fish passage of target species and life stages at the fish passage design flows (Q_L to Q_H). When deciding the upper flow beyond which passage will be impeded, the guiding principle should be that if the reach would have been passable by fish in the absence of the weir, then passage should not be impeded by the weir's presence.

In the absence of a published relationship for the species of interest, a rule-of-thumb of providing a continuous pathway with water velocities ≤0.3 m s^{-1} has been used to guide culvert design. This can also be applied to velocities on the downstream face of the weir to allow passage of most native species (Stevenson and Baker 2009). It is worth noting that velocities above 1.0 m s^{-1} are unlikely to allow fish passage.

Once the maximum passable water velocity has been determined, hydraulic design equations should be used to determine the slope at which this velocity occurs over the fish passage design flow range taking into consideration weir geometry (i.e., width, shape of the downstream face, substrate on the downstream face). For a given head drop, the slope will then determine the length of the weir. In the case that an acceptable velocity can only be achieved with a length that is insurmountable by fish, then two possibilities exist for providing fish passage. Where the site allows, it may be possible to build two shorter weirs, thus halving the head drop and providing a resting pool for fish to recuperate in between the two weirs. Alternately a fish pass, e.g., a partial width rock ramp fishway, should be installed as part of the weir structure (see Section 5.5.10).

Once design velocities have been determined, the associated water depths should be calculated to ascertain whether the depth will provide an impediment to passage of the species of interest. The water depth on the downstream face of the weir over the fish passage design flow range should allow swimming for obligate swimming species (i.e., must be greater than the maximum body depth of the fish).

The use of smooth concrete for the downstream weir face should be avoided where practicable. Roughness should be added to the weir face to create a boundary layer suitable for the movement of fish and to help reduce average water velocities (Figure 4-25). A suitable solution would be to cover the weir face with embedded mixed grade rocks of 150 to 200 mm. Rocks should be closely (70–90 mm) and irregularly spaced to create a hydraulically diverse flow structure across the weir (e.g., Figure 4-25). Rocks should be orientated with their longest axis perpendicular to the weir face and be embedded by at least 50% (e.g. Figure 5-17). The widest axis of the rocks should be orientated into the flow. The inclusion of this feature is of high importance for provision of fish passage.

When designing the downstream weir face, several features must be avoided. Vertical weir faces and overhanging/under-cut downstream faces should be avoided, as this also prevents passage of climbing species. There should be no steps or lips on the face of the weir, as this can create nappe flow with higher levels of turbulence and water level discontinuities, making it harder for fish to negotiate (Baudoin et al. 2015). If a vertical face is necessary for the purpose of the weir and the species of interest includes life stages or species that cannot climb, then a fish pass should be constructed as an integral part of the weir. Partial width rock ramp fishways or technical fishways may provide a suitable solution. These structures are discussed in Sections 5.5.3 and 5.5.10 respectively.

Crest design

Broad-crested weir designs are recommended, and sharp crested designs should be avoided. Broad-crested designs reduce the likelihood of nappe flow occurring, which can impede the passage of fish.

The downstream edge of the crest should be rounded rather than sharp, to allow climbing fish to negotiate the top edge and continue upstream (see Appendix B). If the weir crest requires a notch then it should be v-shaped, as this has been found to assist passage of common bullies when compared with semi-circular or rectangular notches (Baker 2003).

Upstream weir face

Recent overseas research on eels has indicated that the slope of the upstream weir face may have an important influence on the behaviour and movement of fish migrating downstream (Silva et al. 2016). The findings of that study suggest that a 30º incline helps to reduce the maximum water velocity upstream of, and passing over, the weir crest, creating improved conditions for downstream passage (see Appendix B). Rounded (Ogee style) weir crests are also recommended to provide a gradual acceleration of water towards the crest (O'Connor et al. 2017b).

Attraction flows

Attraction flows are most important where the weir has an integrated fishway or bypass. Attraction flows should be available over the entire range of flows, to enable fish to find the path that will allow them to pass over the weir. False attraction flows that do not lead fish to the best upstream pathway can provide a major impediment to passage (Harris et al. 2017). Attraction flows should meet the following requirements (O'Connor et al. 2017b):

- No eddies or recirculation.
- The attraction flow is at the upstream limit of migration or focused on a known area where the target fish species have been found to congregate near the upstream limit of migration imposed by the barrier.
- Other flows do not mask the flow attracting fish to the path that will allow them to pass upstream of the weir.

For more detailed guidelines on attraction flows at weirs see O'Connor et al. (2017b).

Ancillary structures

Vertical wing walls at the edge of the weir should be avoided. Sloping wing walls should be used to ensure that under higher flows a low velocity, shallow wetted margin remains available at the edges of the weir that can assist in providing fish passage.

4.7.5 Summary

Minimum design standards for weirs

- ☐ Where practicable use a full width rock ramp fishway as an alternative to a conventional weir for raising headwater levels in a river.
- ☐ The slope of a rock ramp weir should be gentle. A slope of 1:30 is suitable where weakly swimming species such as īnanga and smelt require passage.
- ☐ Rock ramp weirs should create a hydraulically diverse flow environment including low velocity margins and resting areas.
- ☐ All weirs should have a V-shaped lateral profile, sloping up at the banks and providing a low-flow channel in the centre. 5–10° is a suitable slope for the lateral cross-section.
- ☐ The slope of conventional weir designs should be minimised and generally be less than 1:10 for fall heights ≤1 m and less than 1:15 for fall heights 1–4 m.

- ☐ The use of smooth concrete for the downstream weir face should be avoided. Roughness elements should be added to the weir face. A suitable solution would be to cover the weir face with embedded mixed grade rocks 150–200 mm. Rocks should be closely and irregularly spaced to create a hydraulically diverse flow structure across the weir.
- ☐ A continuous low velocity wetted margin should be provided up the weir throughout the fish passage design flow range.
- ☐ Broad-crested weirs are recommended, and the downstream edge of the crest should be rounded.
- ☐ Backwatering of upstream habitats because of the weir should be minimised.

4.8 Tide and flood gates

4.8.1 Overview

Tide and flood gates are used to control tidal or floodwater fluctuations, respectively. Passive tide and flood gates typically act as a one-way door: allowing water to flow downstream, but preventing water from flowing upstream at times when downstream water levels rise due to tidal fluctuation or rainfall events (Figure 4-26).

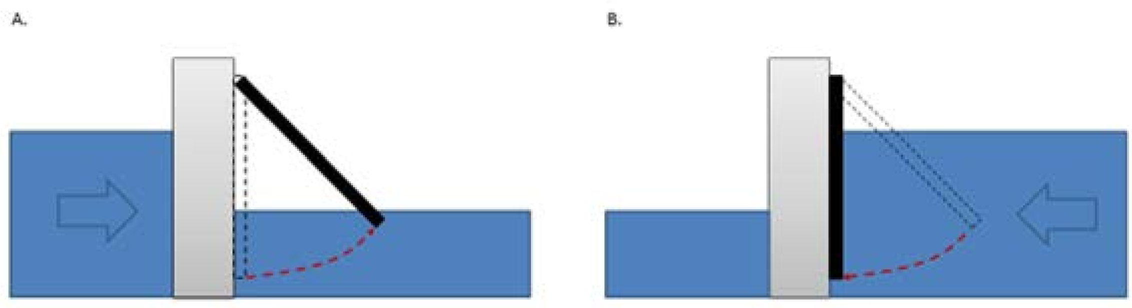


Figure 4-26: Illustration of how passive tide gates work. A. When the water level on the upstream side is higher than downstream, the gate opens. B. When the water level on the downstream side is higher than the upstream side, the gate closes.

All tide and flood gates are considered barriers to fish passage. Furthermore, they degrade upstream ecosystem health by interrupting hydrological exchange and altering water temperature and salinity dynamics (Greene et al. 2012; Franklin and Hodges 2015). Studies that have compared river reaches with tide gates to those without gates have found that passive tide gates reduce the duration of connection between upstream and downstream reaches by up to 86% (Greene et al. 2012; Bocker 2015). When gates are closed, no fish can pass (Doehring et al. 2011a). When gates are closed, they also the interrupt the migratory cues associated with flow and water level variation (Spares et al. 2022).

Many native fish species make obligatory migrations from the sea into river systems as part of their juvenile life stage. Tide gates are often the very first barrier that a fish will face on a migration from the sea into riverine habitat.

The impact of tide gates on access to settlement habitat is, therefore, outsized relative to the number of tide gates that exist in catchments. Differences in fish community composition upstream and downstream of tide gates have been observed in numerous studies (e.g. Boys et al. 2012; Greene et al. 2012; Steinhart et al. 2017; Seifert and Moore 2018). One of the changes in composition that has been observed is a change in the ratio of native to introduced species, with tide gates favouring introduced species (Greene et al. 2012; Scott 2014).

The characteristics of tide and flood gates that present problems for the movement of fish include:

- the duration the gate is open,
- the size of the opening when the gate is open,
- the velocity of water passing through the gate when the gate is open,
- the depth of the opening when the gate is open, and
- the timing of gate opening relative to tidal stage (e.g., flood and ebb).

Studies have quantitatively linked the difference in community composition to the duration of opening of the tide gates (e.g. Greene et al. 2012; Steinhart et al. 2017; Seifert and Moore 2018). Longer opening times lead to a reduction in the difference between upstream and downstream community composition (Greene et al. 2012; Steinhart et al. 2017; Seifert and Moore 2018). Longer duration opening of tide gates has been positively correlated with a higher ratio of native to introduced species (Greene et al. 2012). The opening size of a tide gate has been linked to avoidance behaviour; larger opening sizes corelate with less avoidance behaviour. Larger opening sizes also decrease the velocity through the gate; high velocities have been found to limit passage (Alcott et al. 2021). Different fish species swim at different depths (Bretsch and Allen 2006). The depth of the gate opening can, therefore, selectively allow or prevent access for different species. For example, a top hinged gate that is open and partially submerged has a greater opening size near the bed than at the water surface. The timing of gate opening is also important, as some juvenile fish species (e.g., eels and whitebait) use the flood tide to move upstream (Creutzberg 1961; McCleave and Kleckner 1982; Bocker 2015). Fish passage is improved by ingoing flow, as occurs on the flood tide (Alcott et al. 2021).

Self-regulating tide and flood gates, sometimes referred to as ‘fish friendly’ gates, rely on a stiffener (e.g., a spring that resists the gate closure), float, or counterweight to control the opening and closing of the gate based on the level of the water surface downstream of the gate, or the difference in water level between upstream to downstream (Figure 4-27). In effect, they hold the gate open for a longer period compared to a standard passive gate design. The effectiveness of self-regulating tide gates from a fish passage perspective is highly variable and is dependent on their operating parameters (Greene et al. 2012; Bocker 2015), but their use would be considered the minimum standard for all new and replacement tide gates. Any stiffener, float or counterweight should provide adequate force to keep the gate open for part of the flood tide.

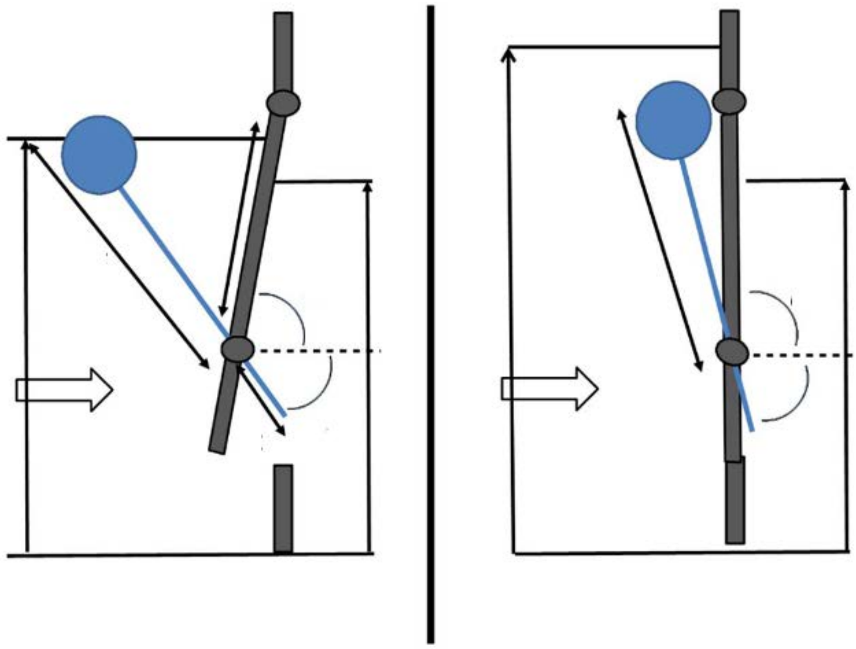


Figure 4-27: Example schematic of a self-regulating tide gate operated by a float on the downstream side of the gate. Modified from Guiot et al. (2020).

In Aotearoa New Zealand, the most common type of self-regulating tide gate is a counter-weighted gate, which uses a wire attached between the bottom of the flap of a top-hinged gate and a counterweight. The system is an off-the-shelf product and, at present, it is not designed to achieve a specified opening duration and/or opening size based on the gate geometry and local water level fluctuations. The length of the wire and the mass of the of counterweight can, however, be adjusted to some extent. In some places these systems have worked very well, for example Bocker (2015) found a 62% increase in opening duration for a lightweight gate that under passive operation was open for 14% of the time. However, in the case of a much heavier gate on a tributary of the lower Waikato, minimal improvement was achieved with this system and the limits of adjustment of the mechanism were reached without a significant increase in opening duration (NIWA, unpub. data). Similarly in the Avoca River, opening duration was only increased by ~17 minutes per tidal cycle (2%) despite the use of lightweight gate materials alongside the modification mechanism (Instream Consulting 2018). Bocker (2015) also recorded a location where the modified tide gate resulted in a small (~6%) improvement in gate opening duration. These case studies highlight the need to consider site-specific conditions and to undertake a full engineering design of a self-regulating gate to ensure that it meets a priori design criteria including the gate opening size and the gate opening duration.

Co-benefits

In addition to facilitating upstream fish passage, co-benefits can be achieved through good tide or flood gate design. When determining the cost-benefit ratio for a particular tide gate design, these are worth considering. Co-benefits include:

- facilitating downstream passage,
- altering salinity and dissolved oxygen upstream of the gates, and
- increasing the size of potential spawning areas.

Longfin and shortfin tuna (eels) have obligatory downstream migrations as adults to spawn at sea. Larval stages of several species (e.g., bullies, galaxiids) also undertake downstream migrations to the ocean. Designing a tide or flood gate to allow upstream passage is likely to also facilitate or increase downstream migration of adult tuna and other fish larvae. In a related species, the European eel (*Anguilla anguilla*), passive tide gates delayed downstream passage by 66 hours (Wright et al. 2014), so the benefit of mitigating this delay is considerable given the energy expenditure associated with the tuna spawning migration. Alterations to dissolved oxygen and salinity upstream of fish-friendly tide gates can have mixed effects (e.g. Franklin and Hodges 2015), but because of the increased exchange of water across the gate the effects are likely to be positive more often than not (e.g. Gordon et al. 2015). Increased salinity upstream of fish-friendly tide gates can increase the area of suitable habitat for estuarine dependent species, although in some cases fish-friendly tide gates made a minimal improvement to habitat for this group of species (Greene et al. 2012). The upstream extent of backwatering due to the design of fish-friendly tide gates can be far-reaching (Guiot et al. 2018). Īnanga spawn on riparian vegetation that is inundated on high tides in tidally-influenced reaches of waterways. Because of this behaviour, the area of potential spawning habitat for this species can be increased by restoring some tidal fluctuation upstream of tide gates (e.g. Jones and Hamilton 2014).

Statutory requirements

Tide and flood gates may be considered dams or flap gates under the NES-F depending on their characteristics. Under the NES-F, new passive flap gates (after 2 September 2020) and their placement, use, alteration, extension, or reconstruction will require resource consent as a non-complying activity.

Resource consent may be required for tide and flood gates that are not passive flap gates. The consenting requirements will vary depending on the design of the tide or flood gate, its location, construction methodology and any other associated activities. Authorisation from DOC will also be required under the FFR83 to determine if a fish facility will be required for a proposed dam or diversion structure.

4.8.2 Design principles

It is extremely challenging to provide effective fish passage at tide and flood gates, thus installation of new gates is strongly discouraged. To date there are limited examples of engineering design of tide and flood gates that allow for fish passage, and the design process is relatively complex (e.g. Guiot et al. 2020; Guiot et al. 2023).

If the installation of a tide or flood gate is unavoidable, there are several design features that can be used to lower the potential impacts on fish movement. Where possible, lightweight materials should be used to decrease the force required to open the gates. The **order of preference for new tide and flood gate installations** is as follows:

1. Avoid installing a gate.
2. Active gate control system (Section 4.8.3).
3. Side-hinged gate with designed stiffener (Section 4.8.4) with specified:
   - opening duration on the flood tide, and
   - opening size.
4. Top-hinged gate with designed stiffener, float, or counterweight (Section 4.8.5) with specified:
   - opening duration on the flood tide, and
   - opening size.
5. Gate always chocked partially open (Section 4.8.6).

It is recommended that an adaptive management approach be applied to any gate system to allow ongoing refinement of the opening regime of the gate to optimise fish passage performance over time. Being able to tune (adjust gate opening angle and duration of gate opening) tidal flap gates during installation/commissioning and over the life of gate, ideally over a relatively wide range, is very useful and is recommended. The design of tide gates is complex, and it does not take much to significantly affect how the gates operate. The ability to adjust opening duration/angle during maintenance, or to mitigate upstream affects later, is valuable.

4.8.3 Active gate control system

In many cases, inundation control is only required under specific circumstances (e.g., during floods for flood gates, or during spring tides for tide gates). Despite this, most passive gate designs remain operational outside these circumstances and close regularly even when not required for flood control purposes. In this situation, active gate designs using automatic electric or hydraulically powered gates, which only close when water levels reach a critical level, can be effective and significantly reduce the impact on fish movements and upstream physical habitat. The use of active gate designs is best practice.

4.8.4 Side-hinged gate with designed stiffener

Side-hinged gates have considerable advantages over top-hinged gates. Side-hinged gates always allow passage at all depths when the gate is open, and velocity will vary only minimally with depth. This gives access to species with different depth preferences and swimming behaviours. A comparison of fish-friendly gates that provided a specified flow and volume of water movement showed that a side-hinged gate was comparable to a top-hinged gate with three times the angle of opening (Guiot et al. 2020). From an engineering design perspective, this means that less force is required to achieve similar hydrological connectivity with a side-hinged design. A spring with a specified stiffness can be installed to provide the required force to delay gate closure (section 4.8.5).

4.8.5 Top-hinged gate with designed stiffener, float, or counterweight

If it is not possible to use a side-hinged gate, then a top-hinged gate with a stiffener, float, or counterweight provides an alternative. A stiffener, float, or counterweight can be used to delay gate closure. The use of lightweight materials for the gate will decrease the force required to delay closure of the gate.

For a top-hinged gate, a float is advantageous for fish passage because it leaves a larger opening than the equivalent stiffener or counterweight (Guiot et al. 2020). For a top-hinged gate, the use of a float rather than a stiffener or counterweight also has the operational advantage of being dependent only on the downstream water level. This means that the operation of the gate can be reliably predicted without considering the hydrological conditions upstream of the gate.

Specifications for a designed stiffener, float, or counterweight

Gate opening time, particularly on the flood tide, and gate opening size are critical parameters for achieving fish passage. We strongly recommend that asset owners require that a new self-regulating (e.g., Section 4.8.4 or 4.8.5) gate is designed to provide a specified[10]:

- duration of opening on the flood tide, and
- opening size.

Analytical solutions that can be used for engineering design of a stiffener for a side-hinged gate are given in Cassan et al. (2018). They provide equations that can be solved in quasi-steady state to determine the required stiffness to give a specified opening size and duration for a given gate geometry and tidal signal (water level as a function of time). Guiot et al. (2020) present more general solutions of a similar form that are applicable to a side-hinged gate with a stiffener or a top-hinged gate with either a stiffener or a float to delay closing, or a block to prevent closing.

4.8.6 Gate chocked partially open at all times

If a gate with a stiffener or counterweight is unachievable or inappropriate for a site, a possible alternative is to design a gate that never fully closes. A block can be placed between the closure and the flap to prevent full closure. We strongly recommend that this setup is properly designed prior to installation so that asset owners and upstream landowners and stakeholders understand the hydrological implications. Solutions can be found in Guiot et al. (2020) for the modelling this type of system.

4.9 Flood pumping stations

4.9.1 Overview

Flood pumping stations are used to mitigate flood risks by draining areas of excess water. Evidence shows that they can have a significant impact on upstream and downstream fish migration (Buysse et al. 2014; Bolland et al. 2019; Baker et al. 2021; Norman et al. 2023). Pump stations can delay and impede fish movements and can also cause high fish mortality due to entrainment and injury of fish when passing through pumps (Vaipuhi Consulting 2017; Bolland et al. 2019; David et al. 2020).

The impacts of pump stations on migratory fishes are known to vary with different site configurations and pump designs. For example, sites with no bypass channel are generally a complete barrier to upstream migration and require that all fish moving downstream pass through the pump. Furthermore, different pump types and models can have significantly different impacts on aquatic

[10] The nature of tidal fluctuations over time and hydrological conditions upstream mean that a designed gate will not meet these specifications 100% of the time, but any requirement may give a minimum percentage of time or a set of conditions for which the duration and opening size specifications are met (e.g., 75% of days in a ~4 week neap-spring tidal cycle that is representative of summer flow conditions).

biota, with some causing close to 100% mortality and others having substantially less impact (Bolland et al. 2019).

There remain significant knowledge gaps regarding the upstream and downstream movements of fishes in the vicinity of pumping stations and how site configuration impacts these movements. Likewise, while there has been notable progress in recent years towards better understanding fish mortality at different pump types in New Zealand, there is a continued need for further information to inform future pump station design and operation.

While there has rightly been a focus on developing solutions to reduce mortality of fish that become entrained in flood pumps (e.g., the development of 'fish friendlier' pumps), it is essential to **take a holistic approach to pumping station design and configuration** that considers the need to provide for both **upstream and downstream movements** of multiple fish species and life stages. This applies for new installations and replacement or upgrade of existing pump stations.

4.9.2 Site configuration

At some flood pumping stations, the only pathway across the stop bank is via the pumps in the downstream direction. This means that upstream movements are prevented, and downstream movements are only provided for when the pump is operating. Where practicable this configuration should be avoided as it is not consistent with the requirements under the NPS-FM to maintain or improve fish migrations unless it is desirable to prevent fish from reaching the upstream habitat. DOC may require the inclusion of a fish facility for all diversion structures.

A common alternative site configuration includes a bypass channel, usually in combination with some form of gravity sluice/flap gate. At these sites, pumps only operate during floods, with water typically draining from the catchment via the sluice/flap gate under the influence of gravity. This configuration can provide a safe upstream and downstream migration pathway, although restrictions on the timing of access may still exist depending on the timing and duration that the bypass is available to fishes (Baker et al. 2021; Mahlum et al. 2022). Bypass channels typically only operate under certain water level conditions and are generally shut during heavy rain events that coincide with directed downstream migration of eels. Under these conditions, fish still have no choice but to pass through the pumps.

Where access to upstream habitats is desirable, the provision of a bypass should be considered an essential component of site design. Use of gravity sluices/flap gates is increasingly recognised as a pragmatic, low-cost solution for providing upstream and downstream passage, but careful consideration must be given to tailoring the operational regime of the flood pump to ensure that the bypass route is preferentially available to migrating fish. This should include consideration of the spatial layout of the bypass entrance relative to the pumps. In most cases, the bypass will be closed at least some of the time during pump operation meaning that effective screening of the pump intake will be required to prevent entrainment of fish in the pump.

For further information on design consideration for flap gates see Section 4.8. For information on best practice for intake screening see Hickford et al. (2023).

4.9.3 Pump design

There are notable differences in the impact of different pump designs and models on fish injury and survival. Evidence suggests that turbine or pump diameter, operating speed, and number of blades can all impact on the amount and severity of impact on entrained fish (van Esch 2012; Buysse et al. 2014; Bolland et al. 2019).

Axial flow pumps, one of the most commonly deployed pump types in New Zealand, are associated with high levels of injury and mortality in eels (Vaipuhi Consulting 2017; Bolland et al. 2019). A study of an 0.85 m diameter, four vane, axial pump in the UK showed 65% direct mortality of European eels (*Anguilla anguilla*) entrained in the pump, with a further 18% of eels suffering mortal wounds (Bolland et al. 2019). Likewise, a study at a 0.37 m, three blade, axial flow pump in New Zealand recorded 100% mortality of eels >600 mm (Vaipuhi Consulting 2017). Mortality of smaller eels appeared to be lower at this site, but there was uncertainty over the number of eels that had passed through the pump versus entered the net from downstream, meaning that no accurate mortality estimate is available for smaller eels. Based on these results, it appears that **standard axial flow pumps are a poor option for achieving downstream fish passage objectives for eels and should be avoided**.

Archimedes screw pumps appear to offer the highest survival and lowest fish injury rates. Buysse et al. (2014) evaluated European eel mortality at small and large conventional screw pumps (where the screw rotates within the casing) and recorded mortality rates of 15–20%. However, newer 'fish friendly' encased Archimedes screw pumps (Figure 4-29) have been demonstrated to provide 100% survival of eels with very low rates and severity of injury overseas and in New Zealand (Vriese 2009; Alicia Williams, WRC, pers. com.). **Where fish passage objectives require high >95% survival of downstream migrating eels, the 'fish friendly' encased Archimedes screw pumps are likely the only option currently available to achieve this objective (Case Study 3)**.

Case Study 3: Mangawhero encased Archimedes screw pump installation

The initiation and subsequent development of land drainage works and flood protection infrastructure is intimately linked with New Zealand's history of colonisation by European settlers. Activities to drain land were initiated by early settlers in the mid-19th century but formalised by central government under the Soil Conservation and Rivers Control Act 1941. Much of New Zealand's current river control, flood protection, and land drainage infrastructure was implemented under this legislation during the 1950s and 1960s by local Catchment Boards with central government funding (Tonkin + Taylor 2018).

The negative impacts of this infrastructure on the environment, aquatic ecosystems, and iwi values were not considered at the time of its development. However, taonga species are highly vulnerable to the impacts of this infrastructure, with tuna being particularly susceptible to injury and mortality at flood pumps due to their elongated body shape.

Tuna mortality events are regularly observed at flood pumping stations throughout the country during flood events. For example, Chetham and Shortland (2009) describe the waters downstream of the Hikurangi Flood Management Scheme (Wairua River in Northland) as "churning with white mutilated bodies of eels"… and "estimated that each pump kills 100s of kilograms, if not tonnes of eels over a 24-hour period."

Tuna are a taonga species for Waikato-Tainui and are fundamental to their customary practices and identity, e.g., "...the Waikato River, with its tributaries, was the most celebrated in New Zealand for its Paa-tuna and the quantities of eels..." (Watene-Rawiri 2021).

The Waikato region has the greatest number (126) of flood pumping stations in Aotearoa New Zealand. Preliminary work indicates that there is close to 100% mortality of eels >0.6 m that become entrained in the standard axial flow pumps installed at most flood pumping stations in New Zealand (Vaipuhi Consulting 2017), although high mortality rates (>90%) have also been observed for smaller (<0.6 m) individuals (Lake and Williams 2020). Evidence indicates that pump mortality occurs at both the silver and yellow eel phase, with the majority of entrainment at one site occurring outside of the main downstream migration period and coinciding instead with the seasonal increase in foraging activity of yellow eels (Mahlum et al. In review).

Ngāti Te Ata Waiohua have been working alongside the Waikato Regional Council and NIWA to replace an old flood pump with a "fish-friendlier" encased Archimedes screw pump and evaluate its effectiveness. This pump is the first of its kind in Aotearoa New Zealand and has been installed on Ngāti Te Ata Waiohua lands at the Mangawhero pump station in Aka Aka (near Waiuku in the Waikato Region). Preliminary results indicate 100% survival and low injury rates for tuna that pass through the new pump (Alicia Williams, WRC, pers. com.)

Figure 4-28: The new Mangawhero enclosed Archimedes screw pump site near Aka Aka. Photo credit: Paul Franklin.

Where lower levels of survival are deemed acceptable and meet the fish passage objectives, there are a range of other pumps available that are described as ‘fish friendly’. The Bedford SAF90 fish friendly pump (Figure 4-29) has been shown to have a low impact on eels <650 mm (Vis and Kemper 2012), but performance for larger eels (>650 mm), representative of downstream migrants in New Zealand, has not yet been quantified. Likewise, there is limited information available on the performance of the smaller Bedford pumps with respect to fish injury and mortality. An evaluation of a Bedford SAF45 that was carried out at the Orchard Road pumping station, Waikato, showed overall eel survival rates of close to 95%, but around 50% of eels that passed through the pump sustained injuries. 8% of all eels were given high injury scores and at least 16% of large (>800 mm) eels either died or sustained injuries considered too severe to complete their downstream migration (Vaipuhi Consulting 2018). Post-mortem analyses of these eels also showed the presence of internal injuries that were not detected from external visual assessments, suggesting that the rate of injury is higher than reported. Consequently, appropriate caution should be applied to the deployment of these pumps in New Zealand until more information is available on eel survivability.

Most drainage and flood pumps installed in New Zealand are manufactured by MacEwans Pumping Systems. MacEwans have recently developed a prototype ‘fish friendly’ pump suitable for retrofitting to sites that are currently fitted with their standard axial flow pumps. A first prototype has been installed in the Lower Waikato and is being tested during 2024.

Fish friendly Encased Archimedes screw pump
Source: FishFlow Innovations

Bedford fish friendly SAF90
Source: Bedford Pumps

Figure 4-29: Examples of two of the more ‘fish friendly’ flood pumps. The encased Archimedes screw pump (Left) shows promise for minimising impacts on the health of entrained fish. The Bedford pumps are preferred to standard axial flow pumps, but still impart severe injury and mortality on some fish species.

4.10 Stormwater management ponds

Stormwater management ponds/wetlands are designed to reduce downstream flooding and erosion in urban and other highly modified catchments.

Watercourses are protected from the effects of pollutants and contaminants washed from impervious surfaces during rain events, and the sedimentation of watercourses is controlled by allowing suspended solids to settle out in the ponds or wetlands, improving the quality of the water entering the natural stream network.

There are two types of stormwater management ponds used in urban regions: dry detention ponds and wet ponds (or wetlands). Dry detention ponds are generally dry but intercept and detain stormwater during and immediately after a storm event, gradually releasing this water over time. Dry detention ponds function both in terms of improving water quality and the reduction of flooding and erosion downstream of the pond. Dry detention ponds do not provide suitable permanent habitat for fish, given their ephemeral nature, and fish passage should not be provided into these systems. Fish have at times been found to gain access into these systems (Instream Consulting 2022), but as far as practicable, fish should be prevented from accessing these ephemeral habitats to avoid them becoming isolated and stranded.

Wet ponds are the main type of pond used and consist of a permanent pond or a constructed wetland where, except for extreme floods, stormwater flows through at a slow rate. Wet ponds can either be ‘on-line’ in which the outflow enters the natural stream network or ‘off-line’ where the outflow enters the stormwater drainage system.

When creating new stormwater management systems, the recommended best practice is to:

- utilise dry detention ponds, or
- develop an ‘off-line’ wet pond system.

In general, fish should be excluded from these systems due to the poor habitat quality. Only in situations where an ‘off-line’ system is unfeasible should an ‘on-line’ wet pond be constructed. For ‘on-line’ systems, good practice is to design a constructed wetland with water levels controlled by a weir at the outlet. The weir should follow the minimum design standards outlined in Section 4.3. Consideration should be given to the presence and potential proliferation of undesirable species, e.g., koi carp, in these systems, and further spread should be avoided. Guidelines on exclusion barriers are provided in Section 6. Vertical risers are not recommended for water level control as they are prohibitive to swimming and climbing fish passage.

Fish passage remediation at existing structures

5 Fish passage remediation at existing structures

There are many existing instream structures in New Zealand's waterways that impede fish migrations. Overcoming this legacy offers the potential for rapid and significant gains for native aquatic biodiversity. The **following section provides a guide to recommended evidence-based options for remediating fish passage at instream structures**. **Other remediation options may be available (e.g. see evidence syntheses in Appendix I and Appendix J), but best available evidence does not currently support their use in New Zealand.** This section focuses on highlighting the key design principles and evidence base necessary for developing site and structure specific remediation solutions.

5.1 Fish Passage Action Plans

The NPS-FM directs councils to develop 'fish passage action plans'. A fish passage action plan must support achievement of the overarching objective to maintain or improve the passage of desirable fish, and any environmental outcomes and target attribute states relating to the abundance and/or diversity of fish. The action plan must set out a work plan for identifying, assessing, and prioritising structures for remediation, and set out targets for remediation. It must also specify how the ongoing performance of remediated structures will be monitored and evaluated. We recommend referring to the relevant regional fish passage action plan when considering actions to remediate fish passage at any instream structure.

5.1.1 Assessing structures

The first step in developing appropriate remediation strategies for existing structures is to evaluate to what extent and why they are not fulfilling the relevant ecological objectives and performance standards (see Section 3 for more detail on objectives and performance standards). This may be achieved through visual assessments, routine, and/or targeted monitoring (see Section 8 for more information on monitoring). The Fish Passage Assessment Tool (FPAT)[11] is a free to download and user-friendly mobile app endorsed by the Ministry for the Environment that can be used to capture the information required to assess the likely risk to fish passage at a structure. The evaluation must include consideration of downstream barriers. Once a structure is identified as presenting a risk to fish passage, the extent and cause of the failure can be identified (e.g., fish passage success is too low because of high water velocities in the structure) and appropriate remediation options to achieve the fish passage objectives can be determined and implemented.

5.1.2 Prioritising structures

Experience indicates that in the region of 30–50% of existing structures currently impede fish passage due to poor installation or inadequate maintenance (Franklin et al. 2022). This amounts to many thousands of structures across New Zealand that may require remediation to meet legislative requirements. It is, therefore, generally necessary to prioritise structures or catchment areas for remediation action.

There are a range of factors that might influence how structures are prioritised including ecological criteria and economic or practical considerations. Some potential ecological factors that may influence prioritisation of structures for remediation are described in Table 5-1. This list is not intended to be exhaustive but provides an indication of the kind of criteria that are valuable to consider from an ecological perspective. They should be used in combination with other relevant

[11] https://fishpassage.niwa.co.nz/

factors such as community support for the project, other restoration efforts underway/completed in the catchment, landownership, practicalities (e.g., is the site accessible for the plant required to undertake the work), and the cost of undertaking the remediation.

The New Zealand Barrier Assessment and Reporting Tool (BART)[12] is available to help tangata whenua and stakeholders to prioritise instream structures for remediation.

Once a potential fish migration barrier has been identified and prioritised for remediation, the next stage is to set objectives and performance standards for the structure (see Section 3), confirm consenting and permitting requirements (see Section 2), and subsequently identify appropriate remediation options for achieving those objectives and performance standards.

Table 5-1: Examples of some possible ecological prioritisation criteria for fixing instream barriers. Multiple factors may influence the priority of works to restore connectivity. This includes not only ecological criteria such as those presented here, but also economic, social, and logistical criteria. Adapted from Franklin et al. (2014).

Criteria	Explanation
Proximity to coast	Barriers that are closer to the coast not only block access to a greater proportion of upstream habitat, but they also generally block a larger number of fish species.
Potential habitat gain	The greater the total length of accessible river upstream of the barrier, the greater the potential habitat gain.
Habitat quality	Restoring access to higher quality instream habitat should be prioritised over providing access to degraded sites.
Proximity to protected areas	Connection with protected area networks may provide added benefits (e.g., constraints on fishing).
Number of species likely to benefit	Some sites are expected to naturally support a greater number of species than others, e.g., sites at low elevation close to the coast. Sites that are expected to support many species may be of higher priority.
Conservation status of species	Sites expected to support species with a higher conservation status may be of higher priority for restoration of connectivity.
Preventing spread of exotic and invasive species	Maintaining boundaries on the spread of exotic and invasive species may be a desirable outcome of retaining barriers and should also be considered in prioritising restoration actions.
Protects threatened species	Barriers may protect populations of threatened fish species by preventing access to competing species, e.g., trout. Existence and protection of threatened fish populations should also be considered.
Site hydrology	Frequency of structure drown out.

5.2 Planning for remediation

Remediation efforts should be based on a clear understanding of the objectives for the site and wider catchment (see Section 3 for further discussion of objective setting). We recommend a remediation hierarchy (Figure 5-1) and provide a remediation decision support framework to assist

[12] https://shiny.niwa.co.nz/barrier-assessment/

with determining the appropriate management interventions (Figure 5-2). Structure removal should always be considered as the first option and is the preferred solution for maximising fish passage at existing structures (see Section 5.3). Alternatively, replacement with a structure that has been designed to provide for fish passage (see Section 4) will likely offer the most sustainable and effective solution. However, for practical reasons many structures cannot be removed or replaced (at least in the short to medium term), so the addition of new features to existing structures is a more common strategy for enhancing fish passage.

Figure 5-1: Planning for barrier remediation should follow the hierarchy of mitigation. The most effective way of restoring river connectivity is to remove the structure. If this is not possible, the next best option is to replace the existing structure with a better design that provides effective fish passage. The final option is to apply remediation to mitigate the impact of the structure on fish passage.

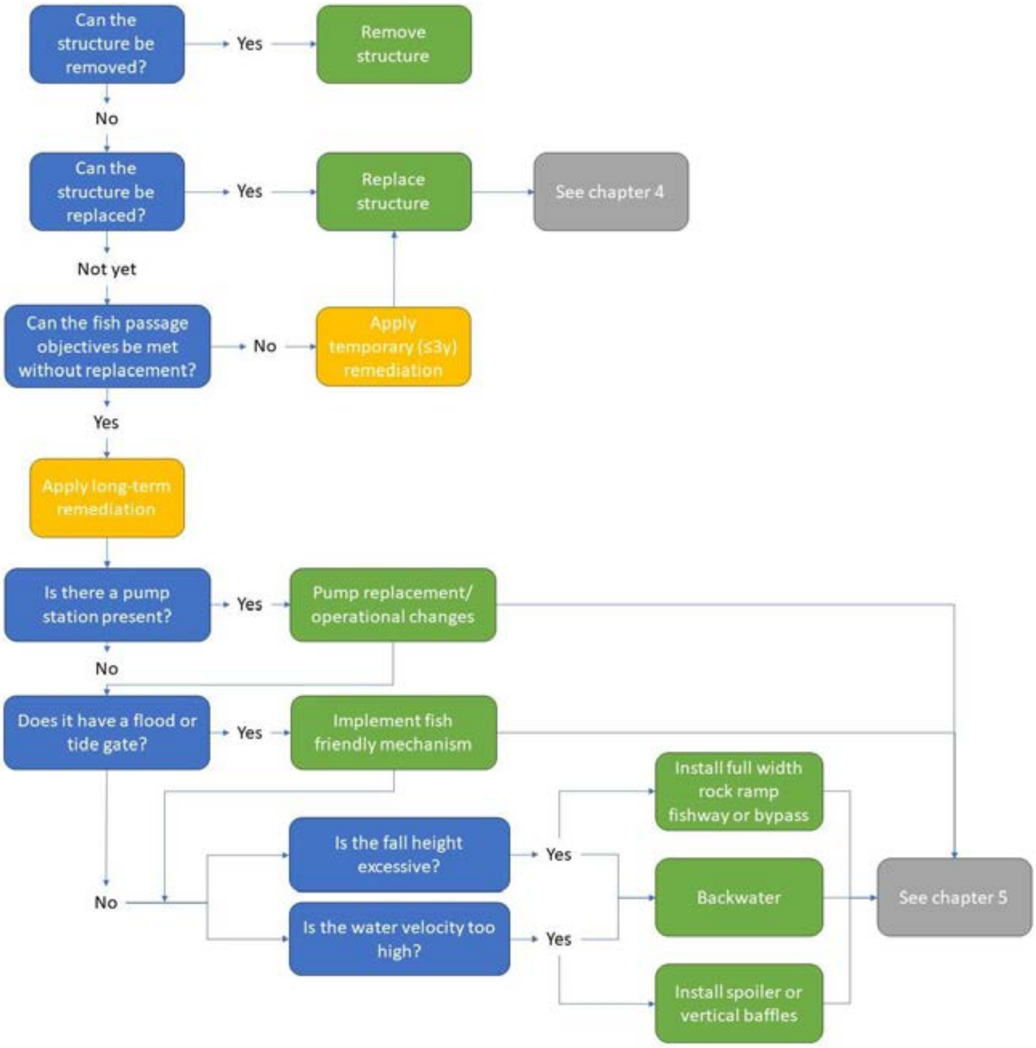


Figure 5-2: Remediation decision support framework.

The remediation options available at a site will be dependent on a multitude of factors including the characteristics of the existing structure (Table 5-2), target species, cost (upfront and lifetime), accessibility, site hydrology, and the reason(s) for reduced fish passage. A combination of remediation options may be required at some sites. It is critical that planning for remediation is transparently and explicitly connected to the objectives and performance measures for the site.

Table 5-2: Common causes of fish passage problems and some possible mitigation solutions.

Common problems	Possible mitigation							
	Removal	Replacement	Backwatering	Ramp fishway	Baffles	Mussel spat ropes	Bypass structures	Fish friendly flap gate
Excessive fall height	Y	Y	Y	Y		Y	Y	
High water velocities	Y	Y	Y		Y	Y	Y	
Insufficient water depth	Y	Y	Y		Y		Y	
Physical blockage	Y	Y		Y			Y	Y

5.2.1 Objectives and performance standards

Establishing clear objectives and performance standards for existing structures provides greater clarity and focus for the fish passage remediation process. It will help to define the design criteria for fish passage remediation at the structure, act as the basis for multiple criteria analysis (see Section 5.2.4) to compare possible solutions and sets the benchmarks against which the effectiveness of the remediation will be measured. The ultimate objective of retrofitting any instream structure should be to achieve unimpeded passage.

5.2.2 Statutory requirements

There is a specific legislative requirement to achieve and maintain unimpeded passage at culverts and fords (i.e., to fulfil the requirements of the FFR83). If this is not feasible (e.g., due to the physical constraints of the existing structure), in the absence of an existing permit, it is necessary to apply to DOC and councils for a permit for exemption or resource consent. This will require a clear justification for any departure from unimpeded access and clearly defined and justified objectives and performance standards for the structure.

Instream structures that dam or divert a natural waterway (e.g., weirs, tide gates, pumping stations) are subject to the requirements of Regulations 43–50 of the Freshwater Fisheries Regulations, in addition to relevant NPS-FM and regional plan rules. It is an offence under the Freshwater Fisheries Regulations to propose to build such structures without dispensation from DOC or an approved fish facility. For any such structure that was built post-1983 and has neither dispensation nor an approved fish facility:

- If you were the builder/authoriser, the Department of Conservation can issue you with a dispensation approving the lack of fish facility, or a requirement to build an approved fish facility.
- If you are not the builder/authoriser (i.e., you are a subsequent landowner) you can get a letter of assurance, or a letter stating that the Department of Conservation would like you to build a fish facility.

Performance standards may be specified as part of the requirements for an approved fish facility and will be important in determining the effectiveness of any fish facility.

5.2.3 Hierarchy of remediation

Remediation inevitably requires compromise and trade-offs between competing values. Achieving unimpeded passage as required by the FFR83 will likely only be achievable through structure removal or replacement and should be the long-term goal. In the short- to medium-term, there are a range of remediation options available that can improve aquatic connectivity. However, the probability of achieving the regulatory goal of unimpeded passage can vary significantly between the options available. As such, when selecting remediation options there is a need for transparent decision-making processes that explicitly document the compromises being made relative to a priori objectives for restoring connectivity.

Based on the evidence syntheses presented in this section and Appendix I and Appendix J, a hierarchy of common remediation options relative to their likelihood of achieving unimpeded passage is presented in Figure 5-3. The effectiveness of any solution will be site- and context-specific, and dependent on appropriate design and installation. Additionally, there remain knowledge gaps around the performance of some solutions. As such, best available information has been used to assign a performance range for each solution that can be used to inform decision-making and be incorporated into a multiple criteria analysis (see Section 5.2.4). Those solutions that sit at the lowest end of the range (i.e., Unlikely to Virtually Certain to Impede) should only be considered as short-term (≤3 year) mitigation options to be used on a temporary basis until long-term solutions can be implemented, because the evidence indicates that they will continue to significantly impede fish movements and, therefore, do not meet regulatory requirements (Figure 5-2).

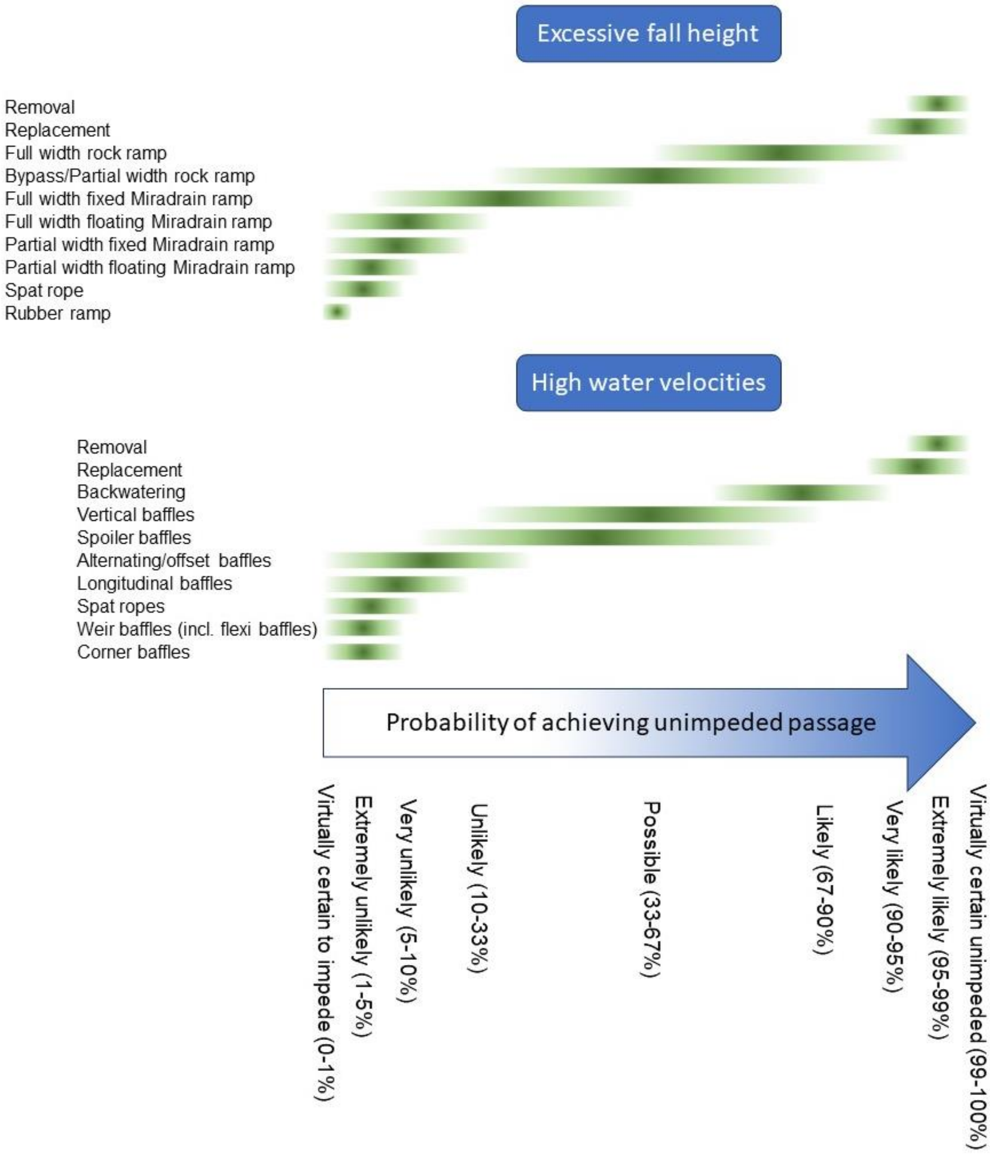


Figure 5-3: Schematic hierarchy of common remediation options for overcoming vertical drops and high water velocities at instream structures.

The hierarchy is based on the evidence synthesis presented in Appendix I and Appendix J. Note that more than one remediation option may be needed as a site.

5.2.4 Multiple Criteria Analysis

Multiple Criteria Analysis (MCA) can be used to compare different remediation options systematically and objectively according to pre-determined criteria and objectives. The MCA criteria should include the fish passage objectives and other critical success factors (e.g., maintenance requirements). A scoring system must also be selected with sufficient range to discern the relative benefits or effects of the various options and weighting may be applied across the criteria. Fatal flaw scores may be included, where an option will not or cannot be achieved. Co-design of the criteria and scoring system with project partners will help to improve the legitimacy of the process. MCA is often undertaken as a group-based assessment since it typically requires input from the full range of specialists, stakeholders and tangata whenua. Utilising a group-based assessment can also help to mitigate potential biases.

Application of MCA offers a transparent way of justifying investment decisions and explicitly quantifying the relative trade-offs between options. It can be good practice to exclude cost in the first round of assessment so that options are not prematurely removed prior to evaluating their relative performance against the selected criteria and objectives.

An example of a hypothetical MCA comparison is presented in Table 5-3. Alongside the ecological objectives, factors such as the operational range of the option, longevity and maintenance requirements of the solution, and logistical complexity of the installation are included. Removal and replacement should generally be included as options in an MCA because sometimes these options may be the more efficient and effective long-term option. In Table 5-3, Option #1 scores highest in terms of its performance against the pre-selected criteria. One of the benefits of MCA is that individual mitigation options can be compared with combined mitigation packages to understand the relative benefits or drawbacks.

Table 5-3: Example of a hypothetical multiple criteria analysis (MCA) comparing different remediation options. Scores range between 1 to 4 with 1 representing a low rating/effectiveness for the criteria and 4 indicating a high level of confidence/effectiveness. Scoring may be qualitative or quantitative but should be objective and free from bias. Criteria should include a priori objectives and performance measures.

Criteria	Option #1	Option #2	Option #3
Whitebait or juvenile fish	✓✓✓✓	✓✓✓	✓✓✓
Adult fish	✓✓✓	✓	✓✓
Swimming fish	✓✓✓✓	✓	✓✓✓
Climbing fish	✓✓✓✓	✓✓✓✓	✓✓✓
Operational flow range	✓✓✓	✓	✓✓✓✓
Entrance attraction efficiency	✓✓✓✓	✓	✓
Aesthetics	✓✓	✓	✓✓✓
Complexity of works	✓	✓✓✓	✓✓
Maintenance needs	✓✓	✓	✓
Lifespan	✓✓✓✓	✓	✓✓

5.3 Barrier removal

The most effective fish passage remediation option available for existing structures is removal. There are many structures in our waterways, both small and large, that are now redundant and no longer serve a purpose. Where such structures are identified, strong consideration should be given to their removal and rehabilitation of the waterway (Case Study 4). Experience has shown that recovery of fish communities and ecosystem processes can be rapid following removal of migration barriers, including large dams (O'Connor et al. 2015b), and so should be prioritised where feasible. Further details on structure removal are included in Section 5.3.

Case Study 4: Kaūpokonui Stream weir removal

Many artificial structures have been installed in Aotearoa New Zealand's waterways over time. Often little consideration was given to the impacts of these structures on ecological or cultural values, e.g., mahinga kai, at the time of their construction. As services and industries have been disestablished and/or aged, these structures may have become disused or degraded. Removal of redundant and ageing structures offers an opportunity to reestablish migratory pathways for fish and restore associated cultural practices and activities.

Iwi and hapū are strong advocates for the recovery of waterways through the removal of instream structures that are impacting their values. For example, Te Korowai o Ngāruahine Trust played a crucial role in the removal of a historical weir on the Kaūpokonui Stream in Taranaki. The 3 m high weir was installed in 1941 (Figure 5-4) to support the now closed Kaūpokonui Dairy Factory, but a weir had been present on the site since around 1900. The weir directly contributed to the loss of non-climbing fish species and reduced numbers of climbing species in upstream habitats.

Figure 5-4: Kaūpokonui Stream weir prior to removal in 2021. Photo credit: Te Korowai o Ngāruahine.

In partnership with Taranaki Regional Council and the Department of Conservation, Te Korowai o Ngāruahine Trust initiated a plan to remove the redundant weir and restore fish communities within the Kaūpokonui Stream. This culminated in the removal of the weir in early 2021 following a ceremony led by Ngāruahine to bless the site before demolition began. This freed up access to approximately 80 km of stream and soon after completion of the works (Figure 5-5), monitoring showed piharau, smelt, īnanga, torrentfish, kōaro, and tuna present upstream of the old weir site. Unfortunately, the weir footing had to be left in place and has subsequently developed into a 1 m high barrier that is once again impeding the upstream movement of fish at the site (Figure 5-5).

Figure 5-5: The site immediately following weir removal (Left) and two years later showing the 1 m high barrier caused by leaving the weir footings in place (Right). Photo credits: Te Korowai o Ngāruahine (left), Finnley Binsbergen (right).

5.4 Structure replacement

Where removal is not a feasible option, replacement with an improved structure that is consistent with the principles of good fish passage design may prove the most cost-effective solution across the lifetime of the structure. Replacement will typically result in more reliable outcomes for passage success than mitigation of the existing structure using the methods described below (e.g., Figure 5-3) and, therefore, will be more likely to achieve fundamental objectives relating to the restoration of fish communities. See Section 4 for guidelines on the design of new structures.

5.5 Remediation

5.5.1 Backwatering

Backwatering can be a simple way to overcome small vertical drops (generally <0.3 m) and help to mitigate shallow water depths and high water velocities upstream. Backwatering involves raising the downstream water level to effectively drown out upstream drops (Figure 5-6). This can also have the effect of slowing down upstream flows and increasing water depths in the structure depending on the extent of the backwatering effect.

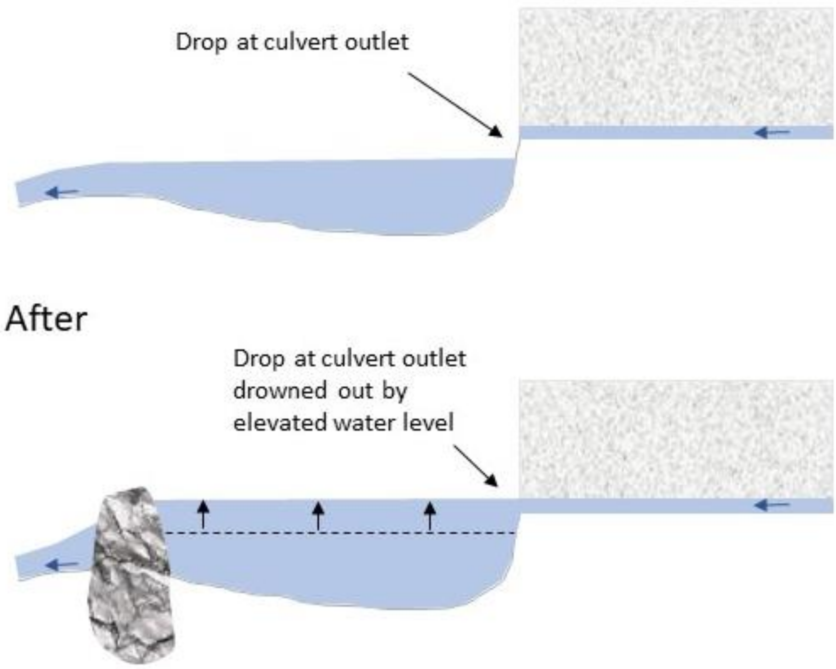


Figure 5-6: Backwatering works by raising the downstream water surface level to inundate the outlet drop. Backwatering can also help to reduce water velocities and increase water depths within a culvert.

The increase in upstream water level can be achieved following the same methods for nature-like rock ramps as described in Section 5.5.3. A rock ridge is formed downstream of the barrier creating a pool with a raised water level. Drops of ≤0.1 m will generally require only a single rock ridge and pool. Larger head drops will likely require additional ridges and pools. The maximum water level difference (see Section 5.5.3 for further details) is critical for determining the number of steps required. It is important that the rocks used to construct the rock ridge are of sufficient size and embedded (60–80%) to ensure stability of the structure. Where fall heights at the structure exceed 0.3 m, a full rock ramp fishway design will likely be required to achieve the backwatering effect.

5.5.2 Fish ramps

Overcoming vertical drops at instream structures is a common challenge for restoring fish passage. Ramp fishways have been widely implemented in New Zealand and overseas for overcoming barriers <2 m in height and have also been used for higher barriers (≤4 m). When ramp fishways are well-designed and maintained they can be a cost-effective and long-term means of significantly improving fish passage success. A variety of ramp fishway designs are in use:

- Rock ramp fishways generally consist of a series of pools created by rock ridges placed below the barrier and connected by continuous water flow. Where practicable, **full width rock ramp fishways are the preferred solution for most sites**.
- Artificial ramps using baffled substrates such as brushes or Miradrain™ have also been used, often at smaller obstructions as short-term fixes.

Full width fishways, which span the full stream width, are strongly preferred over partial width designs and bypasses as they provide greater functionality, have a wider operational range, and exclude the issue of fish needing to find the entrance (a common problem worldwide). Partial-width designs or bypasses (see Section 5.5.10) can be effective but require careful consideration regarding their positioning to ensure fish are attracted to and find the entrance to the ramp (see below).

5.5.3 Rock ramp fishways

Overview

The objective of 'nature-like' rock ramp fishways is to imitate natural stream conditions to disperse the hydraulic head (i.e., vertical drop) over a greater distance, keeping the gradient of the ramp as low as possible. 'Nature-like' rock ramps provide multiple interconnected pathways for fish passage using continuous swimming, or a burst and rest swimming pattern, and typically provide suitable passage conditions and habitat for a variety of species and life-stages over a range of flows.

Full width rock ramp fishways are the optimal design for overcoming low-head barriers (≤1 m) on many river types and are suitable for downstream of perched culverts. They are also practical and will often be the best solution in many situations where the head difference is up to 4 m. The use of 'nature-like' rock ramps has become increasingly common internationally, but uptake of this design in New Zealand has been relatively slow. To be effective, rocks must be carefully configured and structured (Stuart et al. 2024). This will achieve the required structural integrity to minimise the likelihood of failure in flood events.

Design specifications

Rock ramp structures typically take the form of a series of transverse rock ridges, with pool sections between the ridges that act as resting areas for migrating fish (Figure 5-7 to Figure 5-9). Features such as overall gradient, head loss between pools, pool size, minimum water depth and slot width between rocks are all important considerations in the design of these structures. O'Connor et al. (2017b) have provided recommended specifications for rock ramp fishways suitable for small Australian fish species, including īnanga which are widespread in New Zealand. These specifications are summarised in Table 5-4.

Table 5-4: Summary of design specifications for 'nature-like' rock ramp fishways for small-bodied fish. Adapted from O'Connor et al. (2017b).

Design aspect	Specification
Longitudinal gradient	The overall longitudinal slope of the structure should be c.1:20-1:30 for small-bodied (<200 mm) fish but will be dictated by the required pool size and pool to pool head loss.
Functional range	Maintaining a v-shaped cross-section or sloped lateral (bank-to-bank) channel profile will allow the fishway to operate over a greater range of flows than a fishway with a flat lateral profile.
Pool-to-pool head loss	A head loss of 50–100 mm is suitable for small-bodied fish (see below for detailed specifications).
Minimum slot width	The width of the gap between lateral ridge rocks should be ≥100–150 mm.
Pool size	The recommended pool size for a ridge-style rock fishway is generally ≥2 m long to allow dissipation of flow and maintain acceptable turbulence levels.

Design aspect	Specification
Minimum depth	The minimum recommended water depth is 0.4 m in at least 50% of the pool area in a continuous path ascending through the rock ramp (see below for detailed specifications).
Maximum slot water velocity	Maximum water velocity as calculated from the head loss in a vertical slot[13] should be <1.2 m s^{-1}.
Energy dissipation	Turbulence should be minimised, with little 'white' water in the fishway pools. Stream power should be <25 W m^{-3} (calculated as per vertical slot[14]). See below for details on requirements for larger species/life stages.

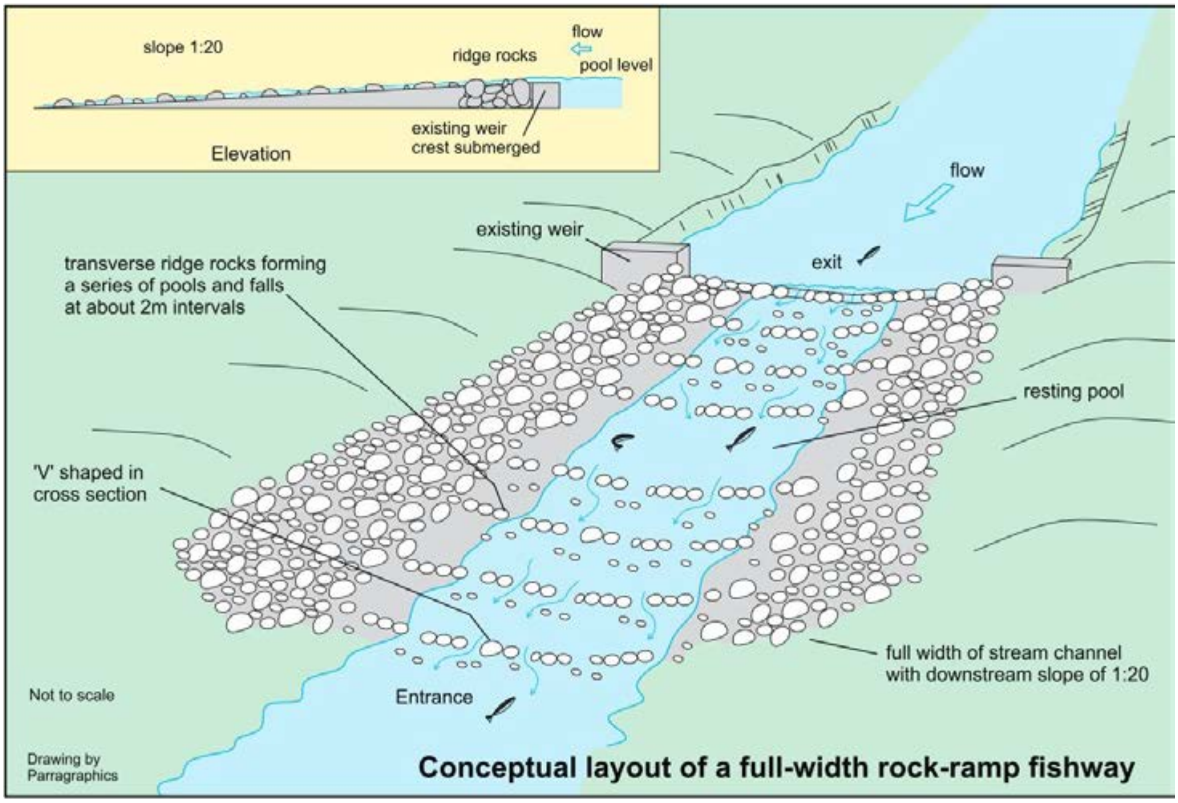


Figure 5-7: Conceptual diagram of full-width rock ramp fishway. Source: Thorncraft and Harris (2000). Credit: NSW Department of Primary Industries.

[13] Calculated as $U = \sqrt{(2g\Delta h)}$, where U = water velocity (m s^{-1}), g = acceleration due to gravity (9.8 m s^{-1}), and Δh = head loss between pools (m).

[14] Calculated as $P = (Q\Delta h\alpha)/V$, where P = Power (W m^{-3}), Q = discharge (m^3 s^{-1}), Δh = head loss between pools (m), α = the weight density of water (9777 N m^{-3} at 25°C), and V = pool volume (m^3).

Figure 5-8: Example of a small low gradient nature-like rock ramp fishway on the Patterson River near Melbourne, Australia. Photo credit: Paul Franklin.

Figure 5-9: Example of a full-width rock ramp fishway in Mill Creek, West Coast. Photo credit: Koen Beets.

Pool size

Pool size should be dictated by two factors:

- Fish size.
- Energy dissipation requirements.

Pools must be large enough to accommodate the largest fish required to pass. A simple rule of thumb is that the minimum pool length should be 5 × the length of the longest fish required to pass. For 'bendy' fish such as eels, the minimum pool length can be calculated as 2.5 × fish length. Because of the widespread distribution of eels in New Zealand's waterways, the size of adult eels (~1 m) will generally dictate the minimum pool length (Figure 5-10).

Pool volume is an important control on turbulent energy within each pool, which in turn has been shown to have a significant effect on passage success. Maximum turbulence thresholds for small-bodied species typical of New Zealand's waterways are low (Table 5-5). Consequently, minimum pool size will often be dictated by the requirement to manage energy dissipation within each pool (Table 5-4; O'Connor et al. (2017b)).

Table 5-5: Design criteria for turbulence in pool-type fishways. Credit: Tim Marsden, Australasian Fish Passage Services.

Minimum fish size (mm)	Maximum turbulence (W m^{-3})
20	25
50	40
100	60

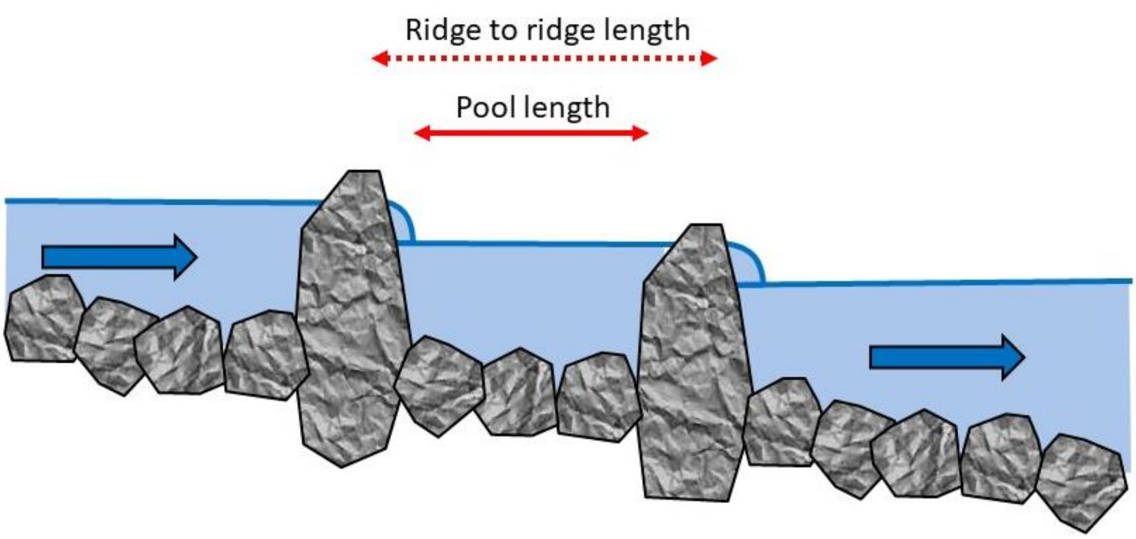


Figure 5-10: Illustration of pool length. Note that pool length is the distance between the rocks, which is less than the ridge to ridge length. It is important to distinguish the two in design specifications.

Pool depth

Pool depth is dictated by the size of fish that must be accommodated within the rock ramp fishway. As fish size increases, the minimum pool depth must increase (Table 5-6).

Table 5-6: Design criteria for pool depth in lateral ridge rock ramp fishways. Credit: Tim Marsden, Australasian Fish Passage Services.

Minimum fish size (mm)	Minimum rock ramp fishway pool depth (m)
<50	0.2
100	0.3
500	0.4

Pool-to-pool head loss

The height of the drop between pools within the rock ramp fishway dictates the maximum water velocity (Figure 5-11; Table 5-4; O'Connor et al. (2017b)). Smaller fish require lower water velocities and, hence, lower drops between pools to successfully pass upstream (Table 5-7). At sites close to the coast where very small juveniles (<50 mm) are prevalent, pool to pool head loss should ideally be kept to a maximum of 50 mm. At sites further inland and/or at higher elevations where the fish community is restricted only to species with climbing abilities, the maximum head loss can be increased to 200 mm.

Table 5-7: Design criteria for head loss in pool-type fishways. Credit: Time Marsden, Australasian Fish Passage Services.

Minimum fish size (mm)	Maximum head loss (mm)
20–50	50
>50	100
>100	150

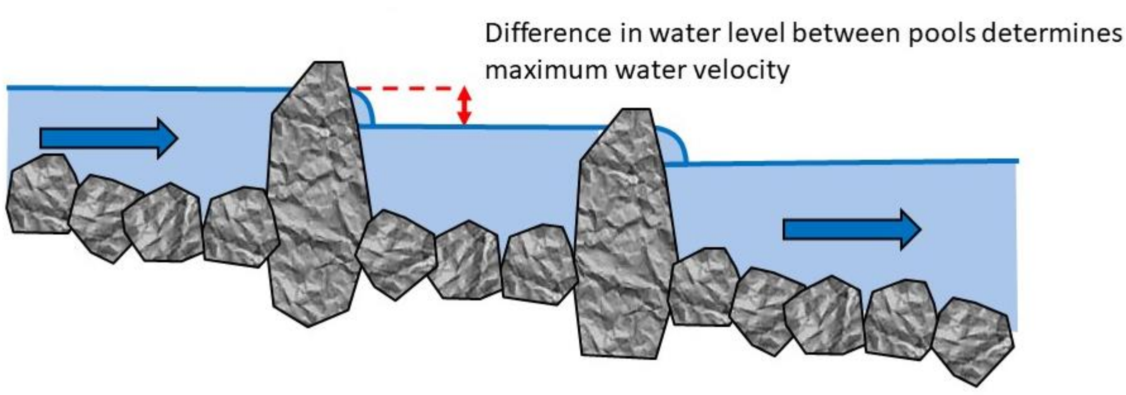


Figure 5-11: Illustration of pool-to-pool head loss.

Ramp length and slope

The length and slope of a rock ramp fishway will be dictated by the maximum water level difference across the structure, the height of the steps between the pools, and the length of the pools.

The maximum water level difference is the maximum difference between the upstream headwater level and downstream tailwater level across the required operating range of the fishway. The ramp length will be equal to:

Ramp length = (maximum water level difference/pool to pool head loss) × pool length

The ramp slope will be equal to:

Ramp slope = $\sin^{-1}$(maximum water level difference/ramp length)

Entrance location

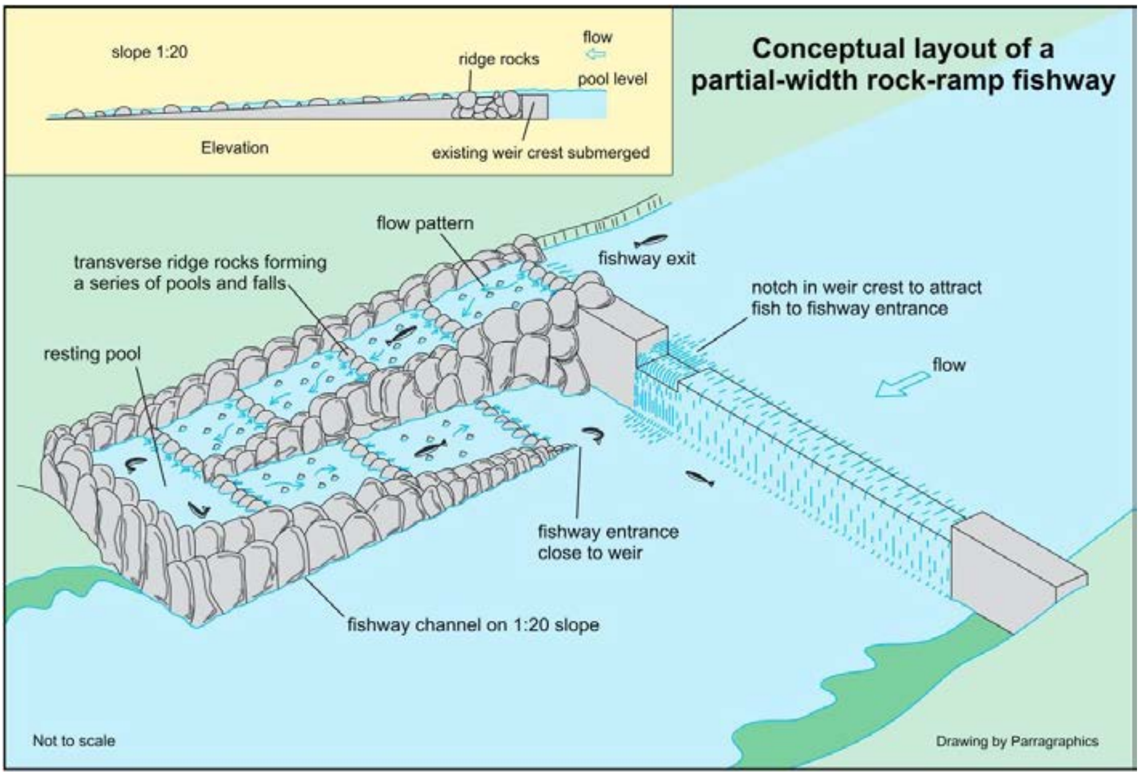
For partial width ramps, siting of the ramp entrance is a critical design aspect. The ramp entrance must be located at the upstream limit of migration which generally occurs at the base of the structure (e.g., Figure 5-12). Failure to locate the entrance correctly will have a significant negative effect on passage efficiency and is one of the primary reasons why full-width ramps are the preferred solution wherever practicable (Stuart et al. 2024).

The upstream limit of migration often varies with flow (or tide). This can create a significant challenge for correctly locating the entrance of a partial width rock ramp (or any other partial width fishway) and is why full width ramps are preferred where practicable.

The key objective in locating the ramp entrance is to place it as close as possible to the upstream limit of migration (Figure 5-13). One solution for reducing this problem has been to install partial-width ramps upstream of the structure to ensure that the ramp entrance can be fully aligned with the upstream limit of migration in a way that avoids fish having to turn into the ramp (Figure 5-13B).

Figure 5-12: Locating the upstream limit of migration (area shaded in red). The upstream limit of migration is the furthest point upstream that fish can move to. The red areas show the upstream limit of location for structures A-E at the time the picture was taken. Note that site E is tidal and the upstream limit of migration moves from the shaded areas to the location indicated by the arrow at high tide.

A



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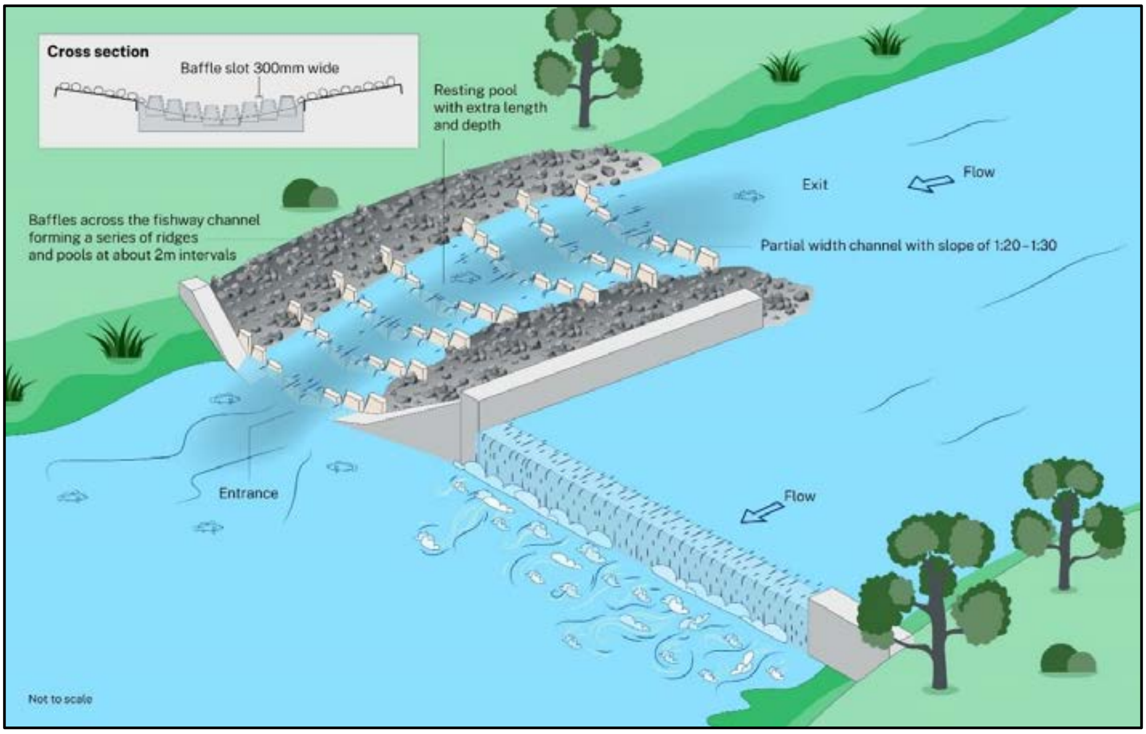


Figure 5-13: Conceptual layout of (A) a partial-width rock ramp installed downstream of a weir and (B) a partial-width rock ramp installed upstream of the weir to improve entrance efficiency. Note that the entrance to the downstream partial-width ramp (A) is located close to the weir at the upstream limit of migration. Example B illustrates the use of pre-cast concrete ridge lines (as opposed to natural rock), which is intended to simplify construction. Credit: NSW Department of Primary Industries.

Exit location

It is vital to locate the exit of the fishway where fish can safely access the river and continue moving upstream (e.g., Figure 5-13). If the exit is located too close to a spillway or within a high velocity area of the downstream flow, fish could be washed back downstream. Where an intake is present, the exit of the fishway should be located well upstream to ensure fish do not entrain into the intake or impinge upon the screens.

Installation

With respect to construction, international guidelines (DVWK 2002; O'Connor et al. 2017b) suggests that rock size is a site-specific decision. General design principles indicate:

- Large diameter rocks embedded a minimum of 60–80% of their longest axis into the fill rock are recommended for the ridge rocks.
- Ridge rocks should generally protrude 0.3 m above the water surface under normal flows and remain protruding from the water surface within the full design operational range.
- The rock ridges will generally be installed with a V-shaped cross-sectional profile.
- The ridge rocks should extend across the total width of the stream and into the banks, and be keyed in.
- Larger infill boulders should be placed to support the protruding ridge rocks.
- It is essential that several layers of graded rock infill are utilised within the structure.
- Geo-fabric material may be used on the rock ramp foundation and upstream face of the ridge rocks to trap fine material and decrease permeability.
- Mixed media fill (20–150 mm) should be augmented with fines to infill interstitial spaces and help ensure the minimum water depth over the ramp is maintained.
- The toe of the ramp should always be secured with a minimum of 2 rows of large rocks, buried to 1 m below bed level and into the banks.

A well-designed ramp should not require grouting (e.g., with concrete) to prevent percolation of water through the structure (Figure 5-14). This avoids problems associated with subsequent settling of the fishway that can result in grouting cracking and being undermined.

All rock ramp installations will need some maintenance over time, but from Australian experience to date this is relatively minor and typically includes debris removal and weed control (Stuart et al. 2024).

Figure 5-14: Example of a newly constructed full width rock ramp fishway in Mill Creek, West Coast. Photo credit: Tim Marsden, Australasian Fish Passage Services.

5.5.4 Concrete rock ramps

Overview

When space is more constrained, concrete rock ramps may be an appropriate solution for overcoming head drops. This option can be fitted downstream of both culverts and weirs and can be a full- (preferred) or partial-width design. Ramps can be fitted directly at the culvert or weir base, or at the base of a receiving pool. The need for a receiving pool will vary depending upon the situation; for example, if the flow downstream of the culvert is to be re-directed from its path through the culvert (e.g., Figure 5-16).

Concrete rock ramps generally take one of two forms:

- **grouted rock ramps** (e.g., Figure 5-15), or
- **formal structural designs** (e.g., Figure 5-16).

A grouted rock ramp can typically be designed to accommodate a wider range of fish species and sizes than a formal structural design due to their flexibility in form.

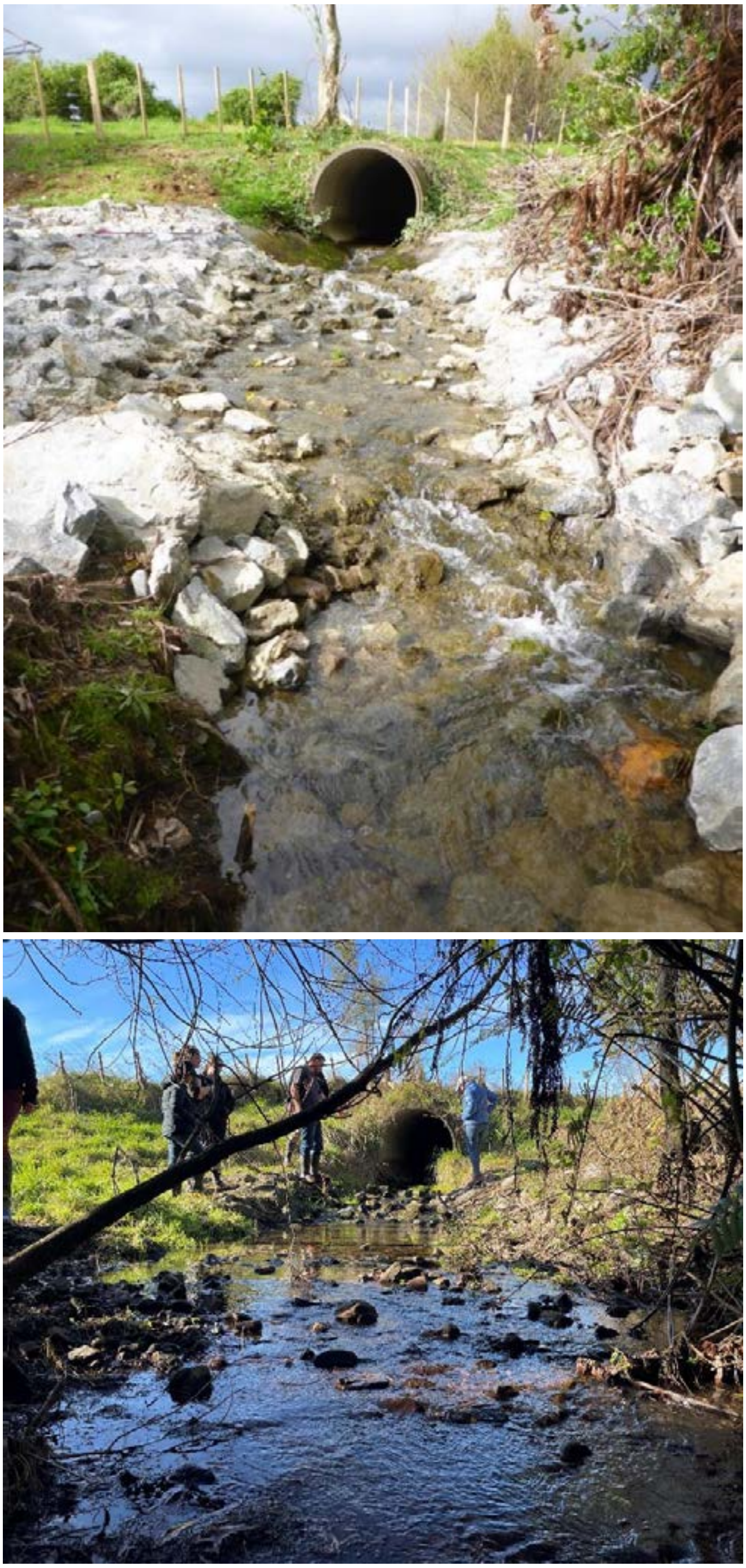


Figure 5-15: An example of a grouted concrete rock ramp below a culvert in the Manawatu-Wanganui region. Top, as built in 2014. Bottom, a decade later in 2024. Concrete is used to prevent water seepage between rocks and is shaped to provide a low flow channel and resting pools to facilitate upstream passage. Photo credit: Cindy Baker.

Figure 5-16: Example of a formal concrete rock ramp below a culvert that is oriented perpendicular to the downstream water body. A receiving pool has been added to the base of the culvert with the ramp directed downstream along the river margin. This provides the foundations for a low gradient sloping ramp. The orientation along the river margin increases the attraction efficacy of the ramp and allows flood flows to be spilled over the receiving pool, perpendicular to the river flow.

Design specifications

Ramp length and slope

For either design, the slope of the ramp should be less than:

- 1:5 for head differences of ≤0.5 m.
- 1:10 for head differences of ≤1.0 m.
- 1:15 for head differences of 1–4 m.

Ramp surface

Mixed grade irregularly shaped rocks (150–200 mm) should be embedded by ≥50%, with the longitudinal axis perpendicular to the ramp surface and the widest part of the stone facing into the flow (Figure 5-16 and Figure 5-17). The rocks should be arranged haphazardly (as opposed to in uniform lines). Spacing of 70–90 mm between rocks should be suitable for most juvenile fish. On steeper gradient ramps, spacing may have to be closer to maintain lower water velocities, although it is useful to have varying spacings to accommodate different fish species and sizes. Ramps should be angled laterally or created with a V-shaped cross-section to provide a range of water depths that taper to a shallow wetted margin (Figure 5-18). This will provide low water velocities along the margins of the ramp for swimming fish and a wetted margin for climbing species. It is essential that the width of the ramp provides a wetted margin throughout the fish passage design flow range.

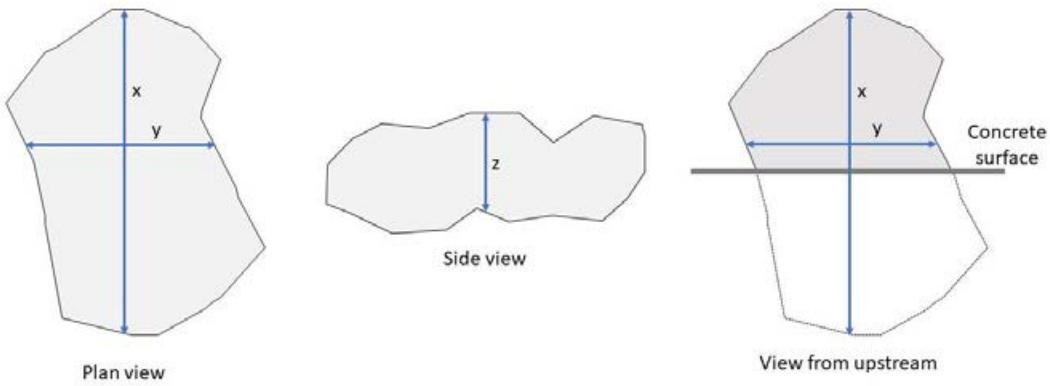


Figure 5-17: Rocks should be embedded into the concrete with the longitudinal axis perpendicular to the concrete surface with the widest part of the stone facing into the flow.

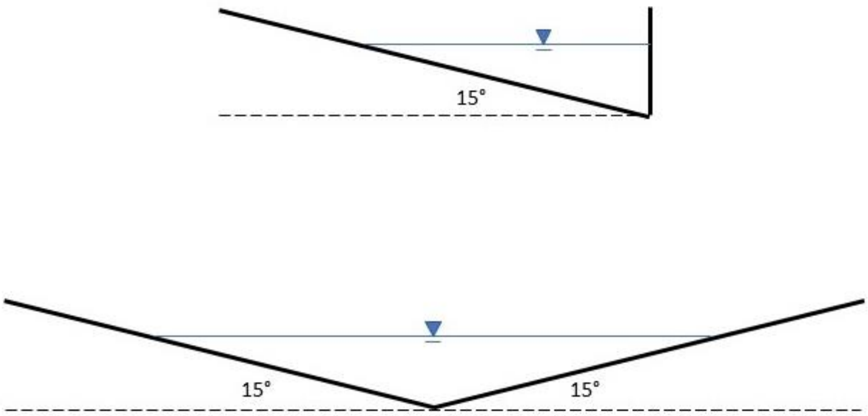


Figure 5-18: Transverse cross-section of a tilted (top) and V-shaped (bottom) ramp showing the lateral tilts that provided a range of water depths tapering to a low velocity wetted margin at the water's edge.

Receiving pool

Utilising a receiving pool before the ramp (i.e., at the downstream base of the culvert) will help provide passage at all flows, as in high flow events some water can flow over the edge of the pool away from the ramp, providing a spillway for excess water. T

This also protects the ramp from damage during flood flows (see Figure 5-16). Any receiving pool should be twice the width of the structure and ramp outlet to provide low velocity margins to aid swimming fish passage. Pool depth will depend upon the flows experienced through the culvert but should be at least 0.3 m. Deeper pools are desirable as they increase energy dissipation and reduce turbulence (see Section 5.5.3 for further discussion of energy dissipation). In cases where the culvert occurs at a stream confluence and flows out into the mainstem perpendicular (or at an angle) to the flow, the ramp should be positioned along the bank and parallel to the mainstem channel (Figure 5-16).

Entrance location

For partial width ramps, siting of the ramp entrance is again a critical design aspect. The ramp entrance must be located at the upstream limit of migration (see Section 5.5.3 for further details). Failure to locate the entrance correctly will have a significant negative effect on passage efficiency and is one of the primary reasons why full-width ramps are the preferred solution wherever practicable.

Installation

Grouted rock ramps take a more natural form where concrete is used as grouting for a rock ramp style fishway (see Figure 5-15). Geo-fabric material can be used on the foundation of the ramp, with mixed grade rocks and boulders used to create the primary channel form. Concrete is then used as an infill to prevent water seepage between the rocks and to form the desired channel shape in the ramp. This should include the provision of resting pools and must include a low-flow channel. The average ramp gradient must adhere to that specified above, but the overall cross-sectional profile will vary contingent upon boulder size and placement. Rocks should remain protruding above the concrete surface to provide the appropriate baffling effect to reduce water velocities and provide low velocity refuge areas. It is also important to ensure the foundations are secure and that water does not seep through the ramp to avoid undermining the structure and flows on the ramp do not dry up. Protection of the toe of the ramp is also important to avoid undermining and maintain the stability of the structure. Installation of large boulders and creation of a receiving pool can be effective ways of providing protection and dissipating energy.

Formal structural designs typically involve constructing a concrete ramp into which rocks are embedded (Figure 5-16). Ramp gradient and cross-sectional profile will be specified in the detailed design and should include specification of measurement tolerances. It is important to ensure that rocks are embedded ≥50% into the ramp surface. Where stream bed or debris load is high, the rocks on the ramp surface should be embedded by ≥75% to reduce the likelihood of displacement following impact. Resting pools should be incorporated into longer ramps to provide resting areas for fish moving up the fishway. The guidelines for pool sizing in Section 5.5.3 can be used.

5.5.5 Artificial substrate ramps

Overview

In New Zealand, a range of artificial substrate ramps have been tested as the basis of designing a cost-effective solution for overcoming low-head vertical drops, for example downstream of perched culverts (see evidence synthesis in Appendix I).

Based on the results of studies by Baker and Boubée (2006), Doehring et al. (2012), Baker (2014), Jellyman et al. (2016), Fake (2018), and Franklin et al. (2021), there is clear evidence that ramp substrate, length and slope, and the provision of wetted margins, are all important considerations in artificial ramp designs. In general, these ramps should be considered as short-term/temporary mitigation that can be deployed prior to more effective mitigation being deployed (see Section 5.2.3).

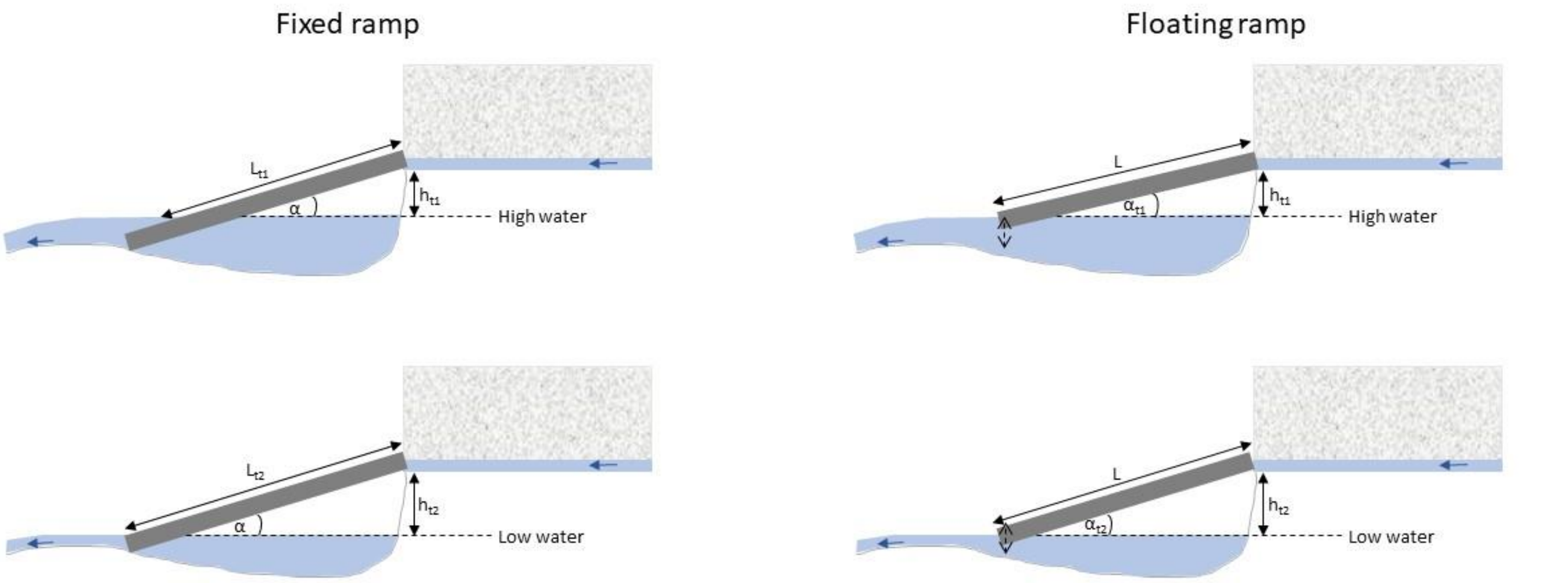
Design specifications

Ramp length and slope

Based on the current evidence base, and presuming a Miradrain™ type substrate, we recommend that maximum ramp slope (α_{max}) should be 15° and ramp length at α_{max} should not exceed 1.5 m (Figure 5-19). This is based on the significant reduction in passage success observed at higher slopes and/or ramp lengths across all studies. Evidence suggests that slopes of closer to 5˚ may be required to achieve high passage rates over ramps of up to 3 m, but slopes of c. 2–3˚ may provide passage up to distances of c. 7 m (Hicks et al. 2008; Baker 2014).

A ramp of 1.5 m at a slope of 15° corresponds to a fall height (h) of 0.39 m, while a 3 m ramp at a 5˚ slope corresponds to h = 0.26 m and for a 7 m ramp at 3˚ h = 0.37 m (Figure 5-19). As such, **artificial substrate ramps are most suited to short-term mitigation in situations where the maximum fall height (h_{max}) is up to 0.4 m**. For fall heights >0.4 m or long-term mitigation, rock ramp designs should be used or a trade-off will have to be made against fish passage efficiency.

Fixed ramps are considered to have a higher passage efficiency across a wider range of species and life stages compared to floating ramp designs (see Appendix I). However, floating ramps can provide some effective short-term mitigation for some species (Fake 2018). An important design consideration for floating ramps is that the slope will vary with the downstream water level and so passage rates are likely to be variable over time as the ramp moves (Figure 5-19). Similarly, the effective ramp length of fixed ramps will vary over time with differing tailwater levels. Ramp gradient (α) and ramp length (L) will be greatest for floating and fixed ramps, respectively, under low tailwater levels. Consequently, ramp installation should be designed to ensure the ramp is within the optimum operating range (e.g., ramp slope ≤15°) under conditions when the tailwater level is low and the head drop is maximised (h_{max}) (Figure 5-19).



	High water (h_{t1})	Low water (h_{t2})	Design considerations
Ramp angle (α)	α	α	Ramp angle constant $\alpha_{max} = 15^{\circ}$ If L > 1.5m, $\alpha < 15^{\circ}$
Ramp length (L)	L_{t1}	L_{t2}	Ramp length changes with water level If $\alpha = \alpha_{max}$, $L \leq 1.5$m @ minimum tailwater level (h_{max})

	High water (h_{t1})	Low water (h_{t2})	Design considerations
Ramp angle (α)	α_{t1}	α_{t2}	Ramp angle changes with water level $\alpha_{max} = 15^{\circ}$ If L > 1.5m, $\alpha < 15^{\circ}$ $\alpha \leq \alpha_{max}$ @ minimum tailwater level (h_{max})
Ramp length (L)	L	L	Ramp length constant $L \leq 1.5$m @ α_{max}

Based on current best available information h_{max} should not exceed 0.4 m for either fixed or floating ramps

Figure 5-19: Design considerations for fish ramps. For fixed ramps at a given slope, ramp length at the low water level will be a primary control on the effective operating range. For floating ramps, ramp angle will vary with the downstream water level, so ramp angle at the low water level will be a primary control on the effective operating range.

Ramp surface

In all cases, ramps should be designed with baffling that matches the Miradrain™ surface (Figure 5-20, Figure 5-21) and a V-shaped cross-section, or tilted laterally to provide a range of water depths that taper to a wetted margin (Figure 5-18, Figure 5-22). This will provide low water velocities for swimming fish and a wetted margin for climbing species.

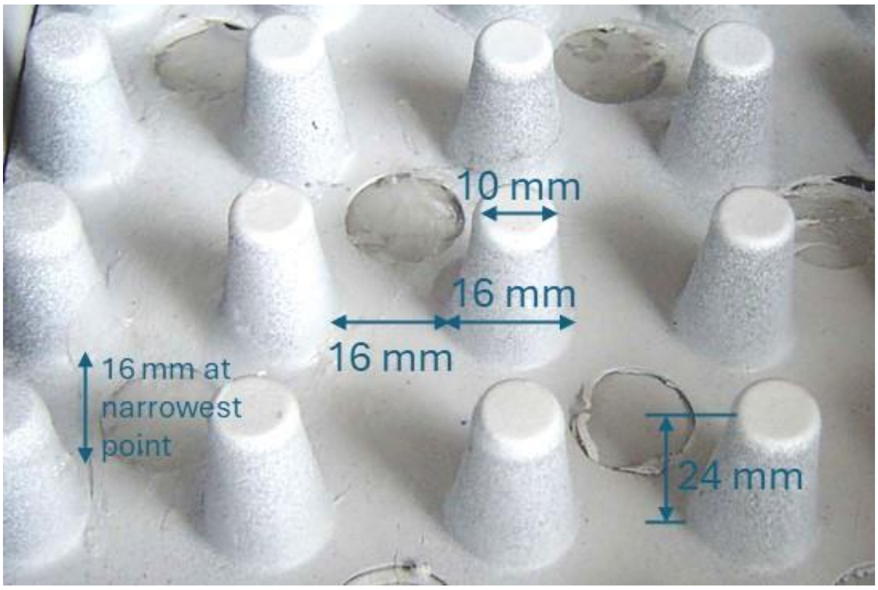


Figure 5-20: Miradrain™ dimensions for application to fish ramps. Miradrain™ consists of parallel rows of tapered raised cups on a flat surface, each cup is 24 mm high, 16 mm wide at the base (tapering to 10 mm wide at the top) and 16 mm apart in both directions.

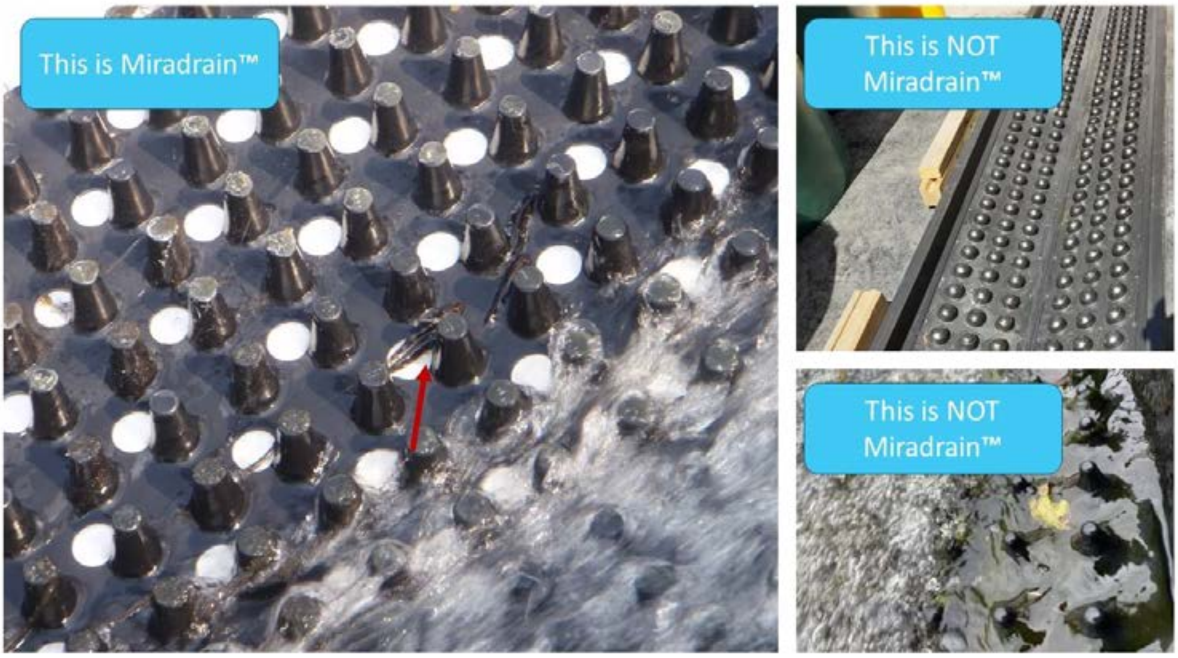


Figure 5-21: Illustration of baffling that matches (left) and do not match (right) the dimensions of Miradrain™. Note that the dimensions of the baffling on the ramps on the right are unsuitable for use in New Zealand. Equivalent baffling was shown to be ineffective in pilot studies carried out by (Baker and Boubée 2006). The red arrow on the lefthand figure indicates a redfin bully using the ramp during experimental trials.

Figure 5-22: Example of an artificial fish ramp installed at a perched culvert in Southland containing the Miradrain™ substrate. Note that a shallow wetted margin is maintained on both sides of the ramp. Photo credit: James Dare.

Ramp width

Full-width ramps are preferable for optimising upstream passage success and essential for long-term mitigation as having 100% of the flow travelling down the ramp under the design flow range will reduce the chance of fish missing the ramp entrance. It is also important that ramps are sized so that the shallow wetted margin is maintained across the fish passage design flow range (e.g., Figure 5-22 versus Figure 5-23; Baker and Boubée (2006)). This means that ramp width should be scaled to match wetted width at the structure. Where wetted width at the structure is greater than the width of a single ramp, installing multiple ramps (if ramps are a fixed width) is likely to improve passage.

Entrance location

Where partial width ramps are used, careful consideration must be given to where the ramp is located and the amount of water that flows down the ramp. Fish will be attracted to the main flow and aggregate at the upstream limit of migration (see Section 5.5.3 for information on the upstream limit of migration). If a small proportion of the flow travels down the ramp and/or the ramp entrance is sited away from the main flow and upstream limit of migration, fish are less likely to be attracted to the ramp entrance and passage performance will be low (Figure 5-23). Fish do tend to move upstream through the lower velocity marginal areas, so locating ramps to the side of the channel may help to improve passage success (Fake 2018; Skyrme 2020).

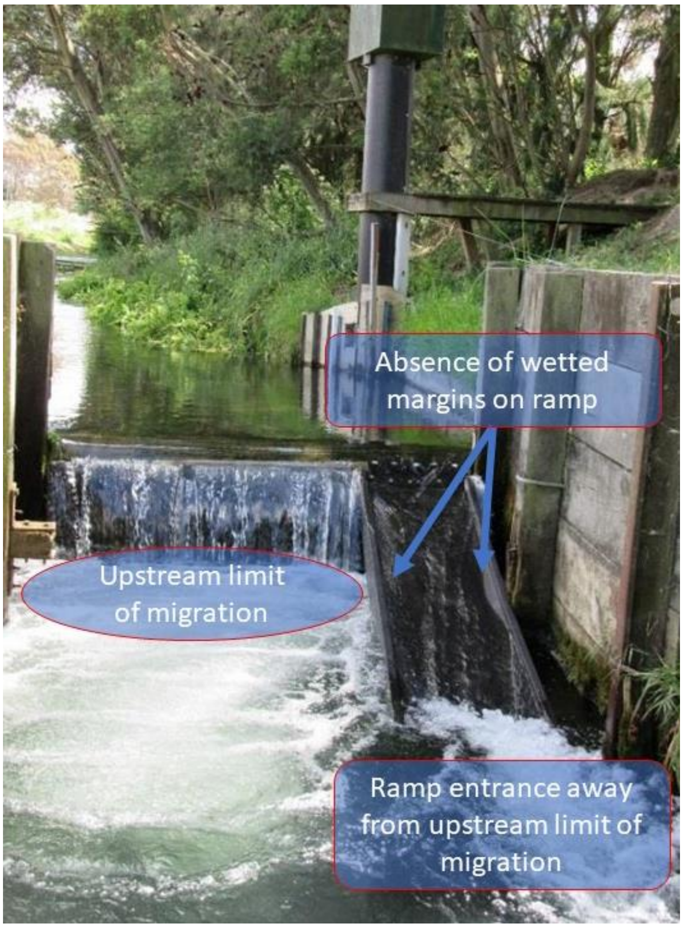


Figure 5-23: Example of a partial-width floating ramp installation. The ramp entrance is located away from the upstream limit of migration which will reduce entrance efficiency. Furthermore, under the pictured flows there is too much flow going down the ramp meaning there are no wetted margins maintained. Photo modified from Fake (2018).

Installation

Ramps should be affixed to the culvert invert approximately 250 mm from the outlet. The type of fixings required will depend on the culvert material, condition, and thickness. Multiple fixings are required to secure the ramp to the culvert invert. It is important to ensure that the ramp fixings do not create a further impediment to fish movements (e.g., an area of high water velocity) at the crest (see Figure 5-24). The downstream end of the ramp should ideally be fixed in place as this improves entrance and passage efficiency for some species when compared to floating ramps ((Baker and Boubée 2006; Fake 2018)). There is currently no evidence that addition of spat ropes to ramps with a Miradrain™ type substrate improves entrance or passage efficiency (see Appendix I). It is, therefore, **recommended that ramps be installed with the laboratory tested Miradrain™ substrate and without the addition of spat ropes**.

Figure 5-24: Artificial substrate ramp installed at a perched culvert on the West Coast. There are multiple ways this installation could be improved. Left: The entrance to the ramp extends past the upstream limit of migration and the ramp is too steep (~35°). The end of the culvert apron is also left unbaffled, which creates a high velocity impediment for any fish able to surmount the ramp. Right: Close-up of the ramp crest showing a high velocity barrier created by the smooth flexible crest used for ramp attachment and the lack of baffling at the crest. Note: this substrate is not Miradrain™ (see Figure 5-20 & Figure 5-21). Photos: Cindy Baker.

5.5.6 Baffles

Overview

A common cause of impeded fish passage at instream structures is water velocities that exceed the swimming capabilities of fish. Baffles have often been used to modify uniform high velocity conditions in culverts or across weirs to improve fish passage success (Macdonald and Davies 2007; Franklin and Bartels 2012; Forty et al. 2016; Amtstaetter et al. 2017; Frankiewicz et al. 2021). Baffles typically comprise plates, blocks or sills that are attached to the culvert base and/or walls, or weir face, in regular patterns with the objective of increasing boundary roughness, reducing water velocity, dissipating energy, developing flow patterns to guide fish, and to create low velocity resting zones for fish.

A comprehensive evidence synthesis on the suitability of different baffle types for facilitating the upstream passage of small-bodied fishes was carried out to inform the recommendations for guidelines at New Zealand structures. The results of the evidence synthesis are presented in Franklin and Baker (In review) and are summarised in Appendix J.

Recommended baffle design for New Zealand

Best available information shows that spoiler baffles and vertical baffles are currently the most effective baffle design for improving passage efficiency and minimising migratory delays through culverts for small-bodied native fishes in New Zealand (Franklin and Baker In review). Recent studies from Australia also suggest that there may be merit in exploring the application of longitudinal baffles for New Zealand species (Watson et al. 2018b), but these have not yet been evaluated here or for pipe culverts. In contrast, the overall weight of evidence indicates that all weir style baffles (including flexi-baffles), and small corner baffles, should not be used in New Zealand because they delay upstream movements (Feurich et al. 2012; Franklin and Baker In review). There is weak evidence that alternating baffles may enhance the passage efficiency of some species (Patchett 2023). Consequently, spoiler baffle and vertical baffle designs are the preferred solution for improving fish passage through culvert barrels in New Zealand (Figure 5-25). If baffles are installed, it is important that robust and defensible outcome monitoring is undertaken to improve the evidence base for future guidance and strengthen our knowledge of what remediation improves fish passage for our New Zealand fish species. For further details on the evidence base underpinning these recommendations please refer to Franklin and Baker (In review) and Appendix J.

Figure 5-25: Example of spoiler baffles (left) and vertical baffles (right) installed inside culverts. Spoiler baffles credit: NIWA. Vertical baffles credit: Tim Marsden, Australasian Fish Passage Services.

Design specification

Baffle sizing and configuration should be adjusted to suit the target fish species, culvert size, and range of flows over which fish must be passed (Rajaratnam et al. 1991; Ead et al. 2002).

Spoiler baffles

Spoiler baffles are appropriate for applications in pipe and box culverts. Stevenson et al. (2008) defined a standardised spoiler baffle configuration for applications in New Zealand that was based on rectangular baffles (0.25 m length, 0.12 m width and 0.12 m height) in a staggered configuration with

0.2 m spacing between rows and 0.12 m spacing between blocks within rows (Figure 5-26). The spacing of the baffles is set to help ensure that fish can use the resting areas created between rows of baffles. A spacing of 0.2 m between rows of baffles will ensure that migratory fish up to 200 mm in size (which will include most adult native fish) are able to fit between rows. This configuration has subsequently been validated in the field as providing effective passage for īnanga and smelt (Macdonald and Davies 2007; Franklin and Bartels 2012).

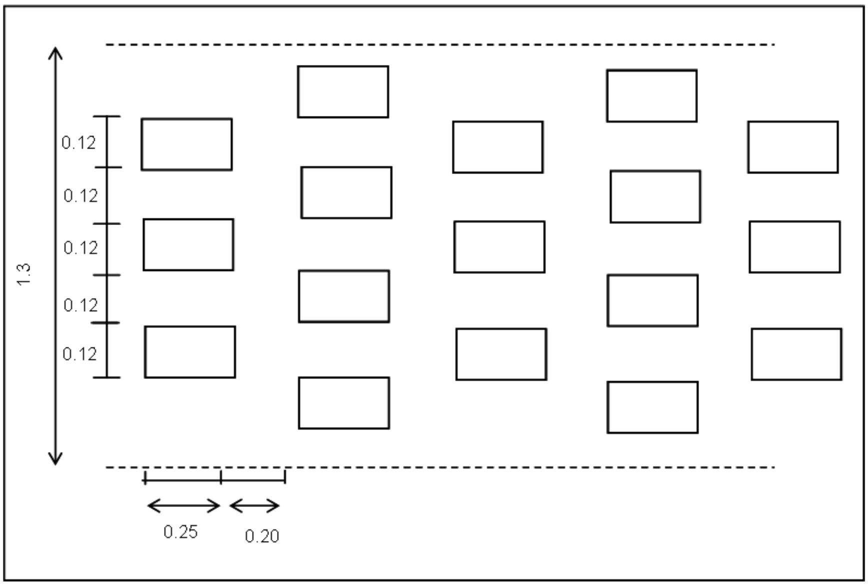


Figure 5-26: Plan view of spoiler baffle arrangement within a 1.3 m diameter culvert. Rectangles represent baffles (0.25 m length, 0.12 m width and 0.12 m height). Dotted lines signify culvert edges, at ⅓ diameter. Rows of baffles are staggered and alternate in rows of three and four baffles. All dimensions are in metres.

Recent work by Magaju et al. (2021) and Magaju et al. (2023) comparing fish behaviour within spoiler baffle arrays with different sizing and spacing has indicated that shorter baffles (0.12 m length) with a spacing of 0.33 m between rows of baffles increases the area of low velocity zones compared to the standard configuration (Figure 5-27). Furthermore, it allows for better development of the wake eddy zones that juvenile galaxiids exploit for resting during upstream migration (Magaju et al. 2023). As yet, this has not been tested at full scale for its overall impact on passage efficiency, but the altered configuration has the potential for increasing passage success while also reducing the negative impacts on flow conveyance arising from the presence of baffles within the culvert compared to the standard spacing.

For culverts with a slope of >2% it may be necessary to adapt the sizing and shape of spoiler baffles to ensure suitable hydraulic conditions are available for fish passage. Stevenson et al. (2008) indicated that smaller baffles (0.12 × 0.12 × 0.12 m) with the same configuration and spacing as the standard baffles (i.e., 0.2 m between rows and 0.12 m between blocks within rows) may be more effective at creating lower water velocities in the culvert barrel than the standard baffle size at a slope of 3%. However, the performance of this configuration is yet to be evaluated with respect to its effectiveness for facilitating fish passage. Consequently, applications outside the standard operating range should be evaluated (see Section 8 and Baker et al. (2024a) for further guidance on monitoring design).

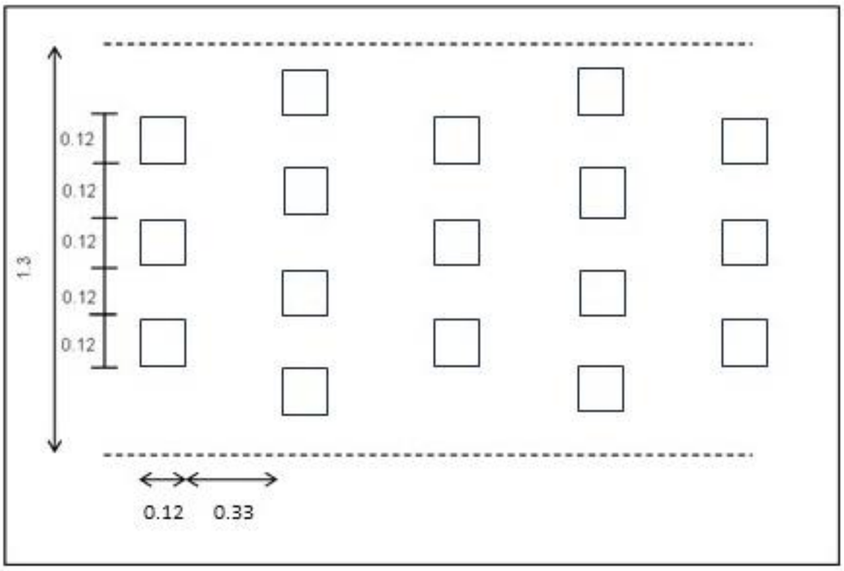


Figure 5-27: Plan view of potential revised spoiler baffle arrangement within a 1.3 m diameter culvert. Rectangles represent baffles (0.12 m length, 0.12 m width and 0.12 m height). Dotted lines signify culvert edges, at ⅓ diameter. Rows of baffles are staggered and alternate in rows of three and four baffles. All dimensions are in metres.

The number of spoiler baffles fitted to a culvert will vary with culvert size, but as a general rule Stevenson et al. (2008) suggested baffles should cover approximately ⅓ of the culvert's internal circumference for pipe culverts or the full width of box culverts. It is essential that they are installed to cover the full length of the culvert.

Vertical baffles

Vertical baffles take the form of individual plates attached to the side of culverts at a pre-defined spacing. The form of vertical baffles is different between pipe and box culverts due to the different geometry of the culverts (Figure 5-28). In box culverts, the baffles are rectangular and are generally sized to span the height of the culvert allowing for a wider operational flow range. For circular culverts, the baffles are curved and generally cover between one quarter and ⅓ of the circumference of the culvert starting from the centre of the culvert base (Figure 5-29).

Presently, the recommended baffle width is 100 mm. Baffles are typically constructed of 6 mm marine grade aluminium or stainless steel. Baffles are installed on the side wall of the culvert at a spacing of 2 × baffle width (i.e., 200 mm) for the first 2 m of the culvert (at the inlet) and 4 × baffle width through the remainder of the culvert barrel (O'Connor et al. (2017a); Figure 5-29). Spacing should be adjusted relative to baffle width. Baffles do not need to be placed along the culvert headwall or tail wall.

Vertical baffles may be installed on one or both sides of a box culvert, but are only installed on one side of a pipe culvert (O'Connor et al. 2017a). For multi-barrel culvert installations, baffles should be installed on the outer most walls of the outer culverts (Figure 5-30). One of the main potential advantages of vertical baffles over spoiler baffles is that they likely have a wider operational flow range because they span a greater water depth within the culvert (O'Connor et al. 2017a). However, they may provide less benefit than spoiler baffles at low flows when water depths are shallow or, for example, on aprons.

Figure 5-28: Examples of vertical baffles installed in a box culvert (left) and pipe culvert (right). Photo credit: Tim Marsden, Australasian Fish Passage Services.

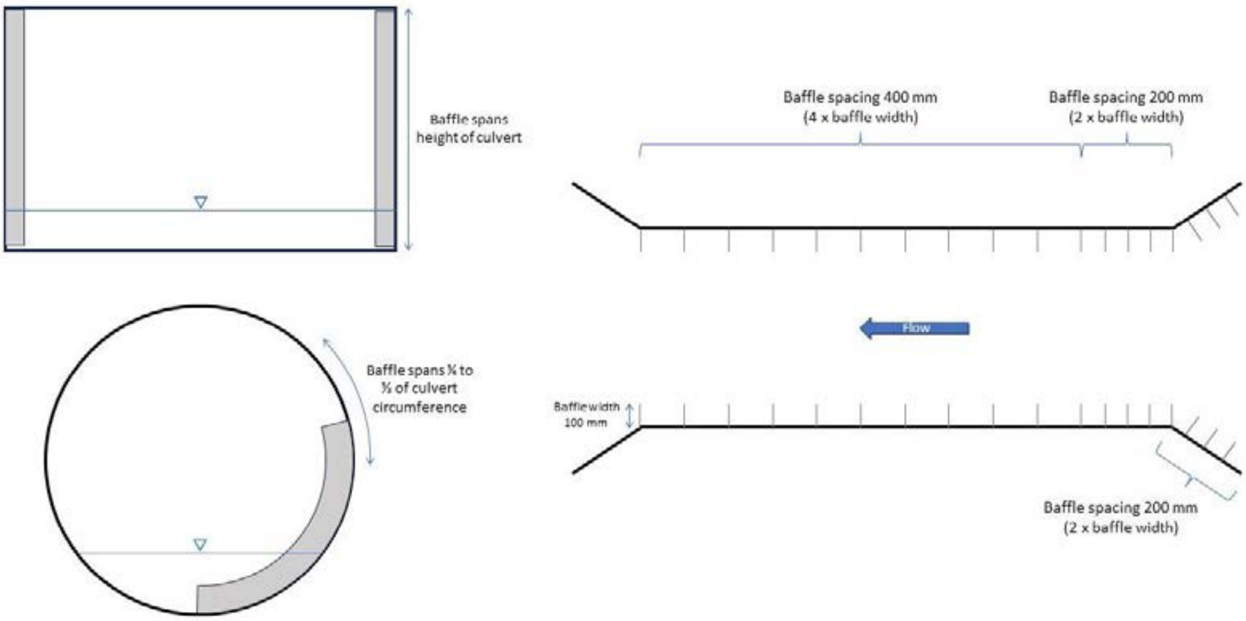


Figure 5-29: View of vertical baffle configuration and spacing within a culvert. Baffle spacing should be scaled with baffle width. Closer spacing is required at the culvert inlet to account for the acceleration of flow that typically occurs as the cross-sectional area is constricted.

Figure 5-30: Example of vertical baffle installation on the outer walls of multi-barrel culverts. Photo credit: Tim Marsden, Australasian Fish Passage Services.

Installation guidelines

There are two main options for fitting spoiler baffles to culverts and other structures (e.g., culvert aprons): addition of individual blocks or installation of moulded sheets of baffles. Individual blocks can be relatively low cost to construct, but are time consuming to install in standardised configurations, particularly for larger culverts. The moulded plastic sheets have the advantage of being quicker and easier to install. However, it is important that the sheets are affixed to the culvert base securely to avoid water flowing under the sheets and causing them to lift and fail. Almost all failures we are aware of have been a result of poor installation with insufficient fixings. Vertical baffles are affixed to the culvert walls individually.

Moulded baffle sheets or vertical baffles can be attached to the base/wall of concrete, plastic or metal culverts using a variety of fixing types. The type of fixing(s) required will depend on the culvert material, condition, and thickness. It will be important to determine the hydraulic impact and impact forces anticipated from bedload or debris at the culvert to ensure the attachment design will work. This will warn if the forces are likely to exceed the strength of the baffle material. A suitably experienced engineer or practitioner should determine the appropriate fixing on a case-by-case basis.

For spoiler baffle sheets, multiple fixings are required on the ends and sides of each baffle sheet or each individual baffle. The first row of spoiler baffles should be attached flush to the end of the pipe at the culvert inlet and it is recommended that the first row of baffles should have the lesser number of baffles (e.g., in a three and four baffle configuration, the first row should only have three baffles). Sheets should be overlapped for installation (50 mm), with the upstream baffle sheet lying over the downstream baffle sheet. Regular maintenance checks should be carried out to remove any accumulation of debris, particularly after high flow events.

For vertical baffles, at least three flanges for attachment are recommended for pipe culverts: one at each end of the baffle and one in the middle. For box culverts, fixings are recommended every 0.5 m, with a minimum of three fixing points per baffle.

Limitations

The primary limitations on the use of baffles within culverts are access for installation and the reduction in conveyance capacity arising from the addition of baffles. It is generally impractical for health and safety reasons to install baffles in culverts <1.2 m Ø. Furthermore, there is often little leeway in terms of spare conveyance capacity in small culverts to accommodate baffles. In these circumstances, there are few options available for restoring fish passage beyond replacing the structure (although see spat ropes below as a possible short-term fix).

The installation of all baffle types will negatively impact flow conveyance due to the reduction in cross-sectional area and increase in friction. It is essential, therefore, that prior to installing baffles within any culvert there is a clear understanding of the conveyance requirements of the culvert and whether the culvert can accommodate the reduced capacity arising from the addition of baffles.

Numerical modelling has indicated that in a 1.3 m Ø pipe culvert at a slope of 1.2%, culvert fullness is reduced relative to a bare culvert by 8% following the addition of the standard sized spoiler baffles (Feurich et al. 2011). Furthermore, modelling has indicated that the influence of baffles on water depth decreases with increasing flow and with increasing relative culvert size (Ead et al. 2002; Stevenson et al. 2008; Feurich et al. 2011). Table 5-8 summarises the results of the modelling described by Stevenson et al. (2008) that characterised changes in culvert fullness following addition of spoiler baffle arrays of the standard dimensions described above in a range of culvert sizes at a slope of 1.2%. Equivalent data at different slopes, or for box culverts and vertical baffles, are not currently available and will have to be determined on a case-by-case basis.

Table 5-8: Changes in culvert capacity at different flows, for bare pipes and for pipes fitted with spoiler baffles. Dimension of spoiler baffles were 0.25 m length × 0.12 m width × 0.12 m height with longitudinal space between baffle of 0.2 m and lateral space 0.12 m. Staggered rows of three and four baffles were modelled for the 1.3 m culvert, rows of six and seven were modelled for the 2 m culvert, rows of 10 and 11 baffles were modelled for the 3 m culvert and rows of 13 and 14 baffles were modelled for the 4 m culvert. Shaded rows indicate that the baffle array was not completely submerged. Reproduced from Stevenson et al. (2008).

Culvert diameter (m)	Discharge ($m^3 s^{-1}$)	Water depth (m)		Fullness of bare culvert	Fullness of culvert with spoilers	Change in culvert fullness
		Bare	With spoiler			
1.3	0.1119	0.146	0.249	11%	19%	8%
1.3	0.2200	0.209	0.314	16%	24%	8%
1.3	0.2750	0.233	0.341	18%	26%	8%
1.3	0.3300	0.26	0.365	20%	28%	8%
2	0.30	0.202	0.326	10%	16%	6%
2	0.55	0.282	0.426	14%	21%	7%
2	1.10	0.410	0.545	20%	27%	7%
2	1.65	0.511	0.655	26%	33%	7%
3	0.75	0.295	0.423	10%	14%	4%

Culvert diameter (m)	Discharge ($m^3 s^{-1}$)	Water depth (m)		Fullness of bare culvert	Fullness of culvert with spoilers	Change in culvert fullness
		Bare	With spoiler			
3	1.50	0.442	0.577	14%	19%	5%
3	3.00	0.636	0.763	21%	25%	4%
3	4.50	0.779	0.925	26%	31%	5%
4	2.00	0.468	0.597	12%	15%	3%
4	4.00	0.687	0.83	17%	21%	4%
4	7.50	0.971	1.077	24%	27%	3%
4	11.00	1.302	1.175	30%	33%	3%

Another concern that is often raised is the potential for baffles to collect debris and cause culvert blockages. To date, we are not aware of any attempt to quantify the risk arising from debris being caught within baffles. The risk is likely to be site- and context-specific, so appropriate site level risk assessments should be undertaken to ascertain whether this is a legitimate concern. Factors that are likely to contribute to this risk include the size of the culvert, the debris load of the stream/river, and the nature of the infrastructure at the site. It might be that installation of individual baffles may be preferable to baffle sheets where there is a high debris load so that if debris accumulations occur, there is less risk of baffle sheets being lifted and adding to the potential for blockage. It is also essential for any baffle deployment that routine maintenance monitoring be undertaken.

Robust biological testing of spoiler and vertical baffles has been largely limited to low slope (≤2%) applications (e.g. Macdonald and Davies 2007; Franklin and Bartels 2012; Marsden 2015; Amtstaetter et al. 2017). However, the hydraulic modelling that has been undertaken suggests that both spoiler and vertical baffles should generate the required reductions in water velocity to facilitate fish passage across a wider range of slopes. One case study of spoiler baffles deployed in a culvert with a 2.6% slope recorded passage efficiency of 70.9% using a catch and release trial (Patchett 2023), which provides a preliminary indication that the standard configuration may be transferable to higher gradients. However, we recommend monitoring be undertaken for any deployments at slopes above 2% until the evidence base is established to confirm the effective operating range.

5.5.7 Spat ropes

Overview

Mussel spat ropes are typically used as a substrate for mussel larvae in marine aquaculture. They comprise of UV stabilised woven polypropylene. Evidence shows that, when deployed correctly, mussel spat ropes can be used to facilitate fish migration at perched culverts and through water velocity barriers but that their efficacy can vary greatly between species (David et al. 2009; David and Hamer 2012; Tonkin et al. 2012; David et al. 2014a; David et al. 2014b). As such, they are only a suitable remediation option under very specific circumstances.

Experience from existing deployments suggests that they are vulnerable to failure and require regular maintenance and replacement. Concerns have also been raised regarding shedding of microplastics from the ropes. **In most circumstances, spat ropes should only be considered a short-term/temporary mitigation option.**

Using spat ropes to improve passage past perched culverts

Laboratory and field trials have demonstrated that juvenile banded kōkopu can surmount vertical spat ropes (David et al. 2009; David and Hamer 2012). Passage efficiency has only been quantified for banded kōkopu at one fall height, with c. 90% of banded kōkopu successfully passing a 0.5 m drop within three hours under experimental laboratory conditions (David et al. 2009). However, field trials demonstrated that some banded kōkopu can pass ropes at least 2.4 m long (David and Hamer 2012). In contrast, no measurable benefit was observed in field trials at the same site for redfin bully, longfin eel, and shortfin eel. It was noted, however, that both species of eel were only present in very low numbers at this site and so these results may not be representative of the potential benefits for these species which are competent climbers at the juvenile life stage (David and Hamer 2012).

Spat ropes will **only improve passage over vertical drops for the juvenile stages of climbing species** (David and Hamer 2012). Consequently, spat ropes are only suitable for use when the objectives and performance metrics for the site are limited to providing passage for juvenile stages of climbing species. At low head drops, high passage efficiency is likely to be achievable for the climbing galaxiid species. It is also likely that similar passage efficiency could be achieved for the two eel species, but this should be confirmed via targeted monitoring or controlled experiments prior to widespread deployment where eels are present. Passage efficiency likely reduces with increasing head height, but this has not been quantified. Efforts should be made to evaluate passage efficiency at sites where spat ropes are installed to overcome larger (>0.5 m) head drops.

In most cases, fish ramps (see Section 5.5.2) should be used rather than spat ropes to overcome vertical drops as fish should not be forced to climb where this is avoidable.

Where the fish passage objectives require providing passage for non-climbing species and/or adult life stages spat ropes will not be a suitable option for remediation.

Use of spat ropes to overcome high water velocities

In smaller culverts (<1.2 m Ø), where access often makes installation of baffles impractical or culvert capacity is insufficient for the addition of baffles, the use of mussel spat ropes has been proposed for facilitating upstream passage of juvenile fish (David et al. 2014b). Trials with small diameter pipes (0.35 m Ø) up to 6 m long showed that the installation of Super Xmas Tree type mussel spat rope could reduce water velocities by around 75% and improve passage success for īnanga, juvenile rainbow trout (*Oncorhynchus mykiss*) and a freshwater shrimp (*Paratya curvirostris*).

Mussel spat ropes offer a practical low-cost method for promoting passage through long, physically inaccessible culverts. Their effectiveness is, however, dependent on correct installation and limited primarily to improving passage for smaller bodied fish (<150 mm). David et al. (2014a) provided guidelines on the appropriate use of mussel spat ropes for facilitating fish passage through culverts. For installation through a culvert, they recommend:

- A **minimum** of two rope lines are used for a 0.5 m diameter culvert, with more necessary for larger culverts.

- Ropes should be installed so that they are **tight and flush with the base of the culvert** through the entire length of the culvert and **not loose at one end** or out of the water (Figure 5-31).
- Ropes are set out to provide '**swimming lanes**' between the ropes (Figure 5-31).
- **Non-looped mussel spat ropes**, e.g., Super Xmas Tree, should be used as looped ropes may be more prone to trapping debris.
- **Rope ends should be melted** during installation to avoid ropes unravelling.

Please refer to David et al. (2014a) for detailed guidance on the installation of mussel spat ropes.

Figure 5-31: Example of good mussel spat rope installation showing fish 'swimming lanes' between ropes. Note that the number of ropes has been scaled up to match the size of the culvert. Photo credit: Bruno David.

A study of the hydrodynamics of spat ropes showed that they create a low-velocity boundary layer (velocities reduced by 50%) of around 2–5 cm, but if ropes are not tied down, they float, which diminishes the hydrodynamic benefit they create and likely reduces their effectiveness for facilitating fish passage (Kozarek and Hernick 2018). As would be expected, the increased roughness created by the ropes resulted in greater water depth. However, the influence of spat ropes on flow boundaries decreases with increasing water depth and decreasing bulk velocity (Kozarek and Hernick 2018). Manning's n was shown to increase from around 0.008 in a flume with no ropes, to 0.015 with a single rope (c. 10% coverage of the flume bottom), to a maximum of around 0.024 with 45% coverage of the flume bottom (4 ropes) (Figure 5-32). Stream substrates can also settle in and around the ropes (due to lower water velocities), which may further increase roughness and reduce conveyance capacity. This should be accounted for in determining whether this solution is fit-for-purpose at a site.

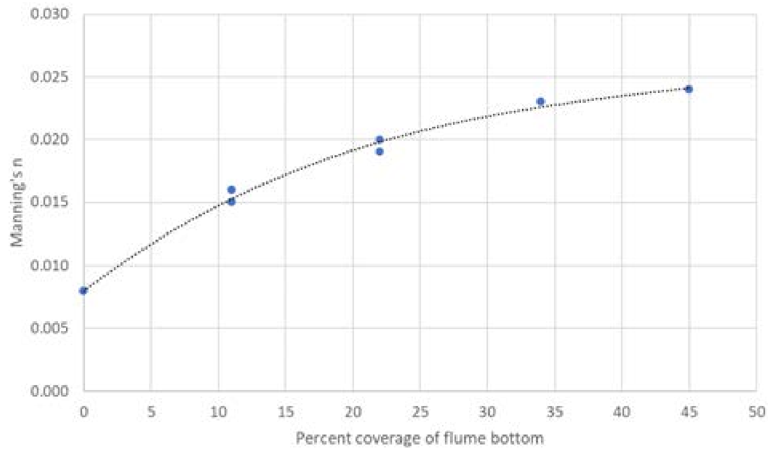


Figure 5-32: Relationship between coverage of spat ropes and Manning's n in a square flume. Source: Kozarek and Hernick (2018).

Limitations

The primary limitation of using spat ropes is that they will not improve passage for all species and life stages. Their benefits are primarily restricted to juvenile life stages and, where used for overcoming drops, climbing species only (David and Hamer 2012; David et al. 2014b).

Presently, little is known about the durability of the ropes in flowing freshwater environments. There are reports from some sites suggesting that, where installed correctly, they may be durable (10+ year life span), but there are also examples of ropes having snapped, disappeared completely or, particularly where not fixed at the downstream end, being left out of the water (Tonkin + Taylor 2017; Kozarek and Hernick 2018). Spat ropes can also sometimes stretch over time so regular maintenance is required.

Another potential issue that has been highlighted is the trapping of debris within the ropes or around the rope fixings, which creates risks of reducing culvert capacity and/or creating blockages. This risk can be reduced by using non-looped spat ropes and avoiding fixings within the culvert barrel (i.e., only fixing at the culvert inlet and outlet). Some practitioners advocate not fixing the downstream ends of the ropes to reduce the probability of debris being caught. However, the performance of loose ropes for improving fish passage is unknown, but it is likely lower compared to fixed ropes that will provide a stable hydrodynamic boundary layer.

A final issue that has emerged with the deployment of spat ropes is the risk of microplastic pollution. It is likely that over time, small plastic fragments will become detached from the ropes and released to the environment. This appears to be a greater risk in environments with more abrasive substrates (i.e., gravel rather than silt) that generate greater wear on the ropes (Kozarek and Hernick 2018). Used mussel spat ropes are available from some mussel farms, but well-worn used ropes are likely to be at much greater risk of shedding fibres and so should be avoided. **It is recommended that spat rope is purchased new when being used for fish passage remediation as this will increase the longevity of the rope and help reduce microplastics being shed to the environment**. When cut, the ends of the ropes should be melted to reduce fraying. Ensuring regular maintenance and replacement of installed ropes will also likely reduce the risk of contributing to plastic pollution until a natural fibre alternative can be found.

5.5.8 'Fish friendly' tide and flood gates

Overview

Remediation of passive tide and flood gates with self-regulating mechanisms that delay gate closing can significantly improve the passage of fish at tide and flood gates and can make community composition more like unmodified reaches (Boys et al. 2012; Alcott et al. 2021; Spares et al. 2022). Most self-regulating gate systems are built around a stiffener (e.g., a spring that resists the gate closure), float, or counterweight to control the opening and closing of the gate based on the water surface elevation downstream of the gate, or the difference in water surface elevation from upstream to downstream (Figure 4-27). In effect, they hold the gate open for a longer period compared to a standard passive gate design. The effectiveness of self-regulating tide gates from a fish passage perspective is highly variable and is dependent on their operating parameters (Greene et al. 2012; Bocker 2015), but their use would be considered the minimum standard for all new and replacement tide gates.

Full details of the issues surrounding tide gates are given in the section on new tide and flood gates (Section 4.8), however the characteristics of tide and flood gates that present problems for the movement of fish include:

- the duration of opening of the gate,
- the size of the opening when the gate is open,
- the velocity of water passing through the gate when it is open,
- the depth profile of the opening when the gate is open, and
- the timing of opening of the gate relative to tidal stage (e.g., flood and ebb).

Remediation projects should seek to address these factors and should especially focus on maximising the size of the gate opening and the duration of gate opening, subject to site-specific constraints. Any stiffener, float or counterweight should provide adequate force to keep the gate open for part of the flood tide.

There is generally greater flexibility to alter the operating regime of flood gates, as they are most frequently only required to provide protection under more extreme flow conditions (i.e., high flow events). This means that it should be practicable to maintain the gates in an open state such that bidirectional flow is unimpeded by the gate structure up to the specified design flood protection levels of the gate.

An important practical consideration for remediating tide and flood gates is to engage with landowners and stakeholders early in the project (Franklin and Hodges 2012). These stakeholders need to be informed and reassured that a balance will be struck between land protection and ecological outcomes. It is our experience that positive, early engagement that includes good two-way communication leads to greater social license for remediation initiatives.

Design principles

The design principles for remediation of tide and flood gates are like those for installing new tide and flood gate structures.

It is extremely challenging to provide effective fish passage at tide and flood gates, thus removal of unnecessary gates is strongly encouraged. To date, there are limited examples of tide and flood gates that allow for fish passage, and the design process is relatively complex (e.g. Guiot et al. 2020; Guiot et al. 2023). If the removal of a tide or flood gate is not possible, there are modifications that can be used to lower the impacts on fish movement. Where possible, flaps or gates should be replaced with lightweight materials to decrease the force required to open the gates.

The order of preference for modifications to remediate tide and flood gates is as follows:

1. Remove the gate.
2. Remove a single gate if there are many in parallel.
3. Replace with active (and automated) gate control system.
4. Modify gate with designed stiffener, float, or counterweight with specified:
   - 4.1 opening duration on the flood tide, and
   - 4.2 opening size.
5. Modify management so gate is always chocked partially open.
6. Add orifice to gate.

Remove a single gate in a group of parallel gates

Removal of a single gate in a group of parallel gates always allows some opening and flow and is, therefore, preferable for passage. It does, however, have greater implications on upstream water levels than other options. When choosing which gate to remove, consideration should be given to factors such as bank erosion potential and species' swimming behaviour.

Active gate control system

In many cases, inundation control is only required under specific circumstances (e.g., during floods at flood gates, or during spring tides at tide gates). Despite this, most passive gate designs remain operational outside these circumstances and close regularly even when not required for flood control purposes. In this situation, active gate designs using automatic electric or hydraulically powered gates that operate the gate only when water levels reach a critical elevation can be effective and significantly reduce the impact on fish movements and upstream physical habitat. **The use of active gate designs is best practice**.

Specifications for a designed stiffener, float, or counterweight

For a top-hinged gate, a float is advantageous for fish passage because it leaves a larger opening than the equivalent stiffener or counterweight (Guiot et al. 2020). For a top-hinged gate, the use of a float rather than a stiffener or counterweight also has the operational advantage of being dependent only on the downstream water level. This means that the operation of the gate can be reliably predicted without considering the hydrological conditions upstream of the gate.

Gate opening time, particularly on the flood tide, and gate opening size are critical parameters for achieving fish passage. We strongly recommend that asset owners require that a gate remediation mechanism is designed to provide a specified[15]:

- duration of opening on the flood tide, and
- opening size.

Analytical solutions that can be used for engineering design of a stiffener for a side-hinged tide gate are given in Cassan et al. (2018). They provide equations that can be solved in quasi-steady state to determine the required stiffness to give a specified opening size and duration for a given gate geometry and tidal signal (water level as a function of time). Guiot et al. (2020) present more general solutions of a similar form that are applicable to a side-hinged gate with a stiffener or a top-hinged gate with either a stiffener of a float to delay closing, or a block to prevent closing.

Gate always chocked partially open

If a gate with a stiffener, float or counterweight is unachievable or inappropriate for a site, a possible alternative is to design a gate that never fully closes. A block can be placed between the closure and the flap to prevent full closure. We strongly recommend that this setup is properly designed prior to installation so that asset owners and upstream landowners and stakeholders understand the hydrological implications. Solutions can be found in Guiot et al. (2020) for the modelling this type of system. They also demonstrate a process to determine the effect of a block on upstream water levels (e.g., see Figure 5-33).

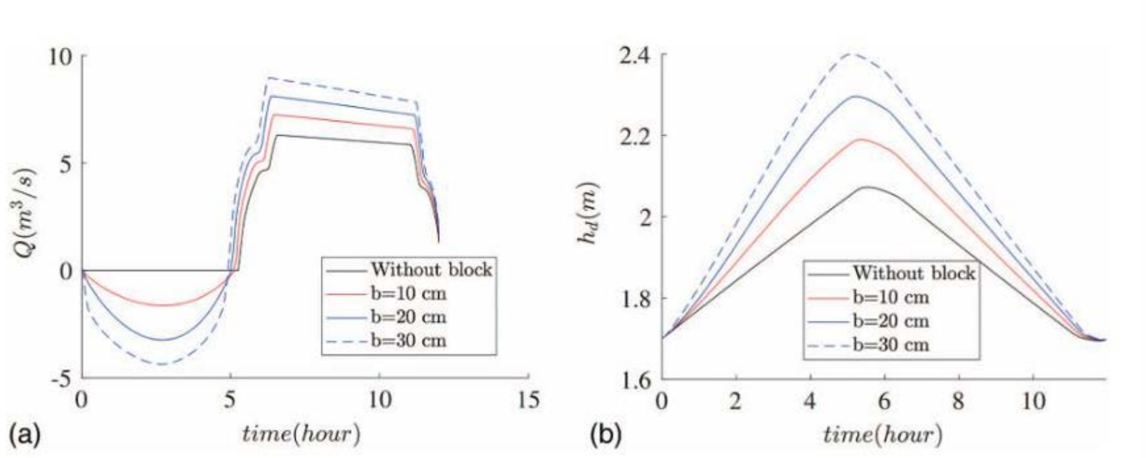


Figure 5-33: The relationship between the size of the block and a) flow through the gate (Q) over time, b) the water level upstream of the gate (h_d) over time. Source: Guiot et al. (2020).

[15] The nature of tidal fluctuations over time and hydrological conditions upstream mean that a designed gate will not meet these specifications 100% of the time, but any requirement may give a minimum percentage of time, or a set of conditions for which the duration and opening size specifications are met (e.g., 75% of days in a ~4 week neap-spring tidal cycle that is representative of summer flow conditions).

Orifice

Published studies have shown questionable efficacy for orifices (Wright et al. 2014; Wright et al. 2016). They do not provide access at all depths and, depending on the positioning of the orifice and the magnitude of tidal fluctuations, they may provide no passage during large parts of the tidal cycle. However, an unpublished study of an orifice installed in the Taranaki Stream (Canterbury) flood gate did show significant upstream passage of several native species. This suggests that orifices warrant further investigation as a possible effective remediation approach (Environment Canterbury unpublished data).

Gate remediation as part of a managed scheme

There has been little research into the optimisation of gate modifications as part of a stream network that is managed by several tide or flood gates. However, some recent work suggests that optimisation of which gates are modified can minimise the number of gates that require modification to achieve desired gate opening sizes and durations for fish passage (Guiot et al. 2023). The backwatering effect arising from modification of the tide gate furthest downstream is quite powerful and can reduce the need to modify other gates in the stream network. This concept has potential for low-lying areas where waterways are managed with a network of tide and flood gates. Guiot et al. (2023) demonstrated a method for using 1D hydraulic modelling to understand the effects of gate modifications on water levels and the operation of other gates in the network.

5.5.9 Flood pumping stations

There are two primary options for remediating pumping stations to enhance fish passage:

1. Replacement of the pump with a 'fish friendly' design.
2. Operational changes.

At sites where no bypass channel is available, pump removal or replacement may be the only option for improving fish survival and passage.

Pump replacement

Many pump designs cause high levels of injury and mortality to fish that become entrained in the pumps. Entrainment is often unavoidable while also maintaining operational service levels. This means the only solution available is to replace the pump with a 'fish friendly' design. Guidelines on fish friendly pump design is provided in Section 4.9.

Operational changes

At sites where a bypass channel is available, it may be possible to introduce operational changes that help to reduce the likelihood of pump entrainment and encourage fish to move via safe pathways. Eels are most active at night, thus avoiding or minimising pump operation between dusk and dawn will reduce the likelihood of fish becoming entrained (Carter et al. 2023; Mahlum et al. In review). It has also been demonstrated that operational management of bypass sluices/gates can be used to artificially simulate high flow events and trigger out migration of eels via the bypass (Bolland et al. 2019; Carter et al. 2023). Where bypass access is controlled by a flap gate, installation of a 'fish friendly' design (see Section 5.5.8 for details) may improve connectivity, particularly for upstream migration.

Trap-and-transfer

In some locations, trap-and-transfer schemes have been implemented as a short to medium-term solution prior to pumping station upgrade for mitigating the impact of flood pumps on fish communities. Further information is available on upstream trap-and-transfers in Section 7.3.3 and on downstream trap-and-transfers in Section 7.4.2.

5.5.10 Bypass structures & fishways

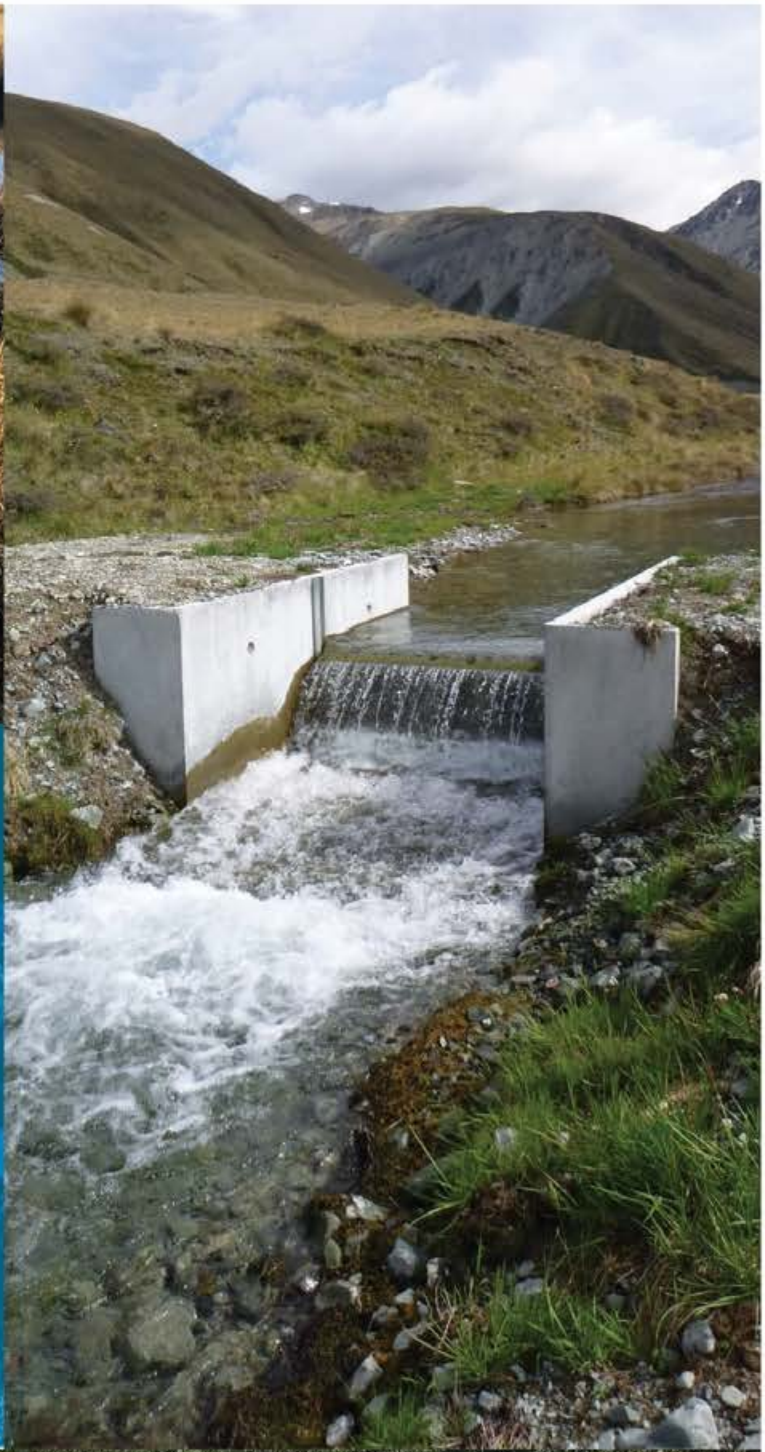
Where fish passage barriers cannot be ameliorated through structural adjustments (e.g., addition of baffles or a rock ramp), bypass structures may be the only effective solution for enhancing fish passage. There are two main types of bypass structure:

1. **Nature-like fishways** mimic natural stream characteristics in a channel that bypasses the barrier. They are suitable for all structure types, but generally require more space than technical fishways. Because they mimic natural stream conditions, they are generally suitable for a wide range of fish species and life stages.

2. **Technical fishways** can take a variety of forms including vertical slot fishways, pool and weir fishways, cone fishways, and Denil passes. There are relatively few examples of effective technical fishways in New Zealand, but they have been widely used internationally.

The effectiveness of bypass structures is highly dependent on their design and layout. They must be sited such that fish can find the bypass entrance and must incorporate conditions that enable fish to successfully traverse the entire length of the bypass channel. It is outside the scope of these guidelines to provide detailed technical design specifications for bypass structures, but some of their key features are summarised in Sections 7.3.4 and 7.3.5. Links to international guides are provided where available and applicable.

Exclusion barriers: a special case for protecting biodiversity



6 Exclusion barriers: a special case for protecting native biodiversity

6.1 What are exclusion barriers?

While providing unimpeded fish passage is advantageous for most fish, some of our native freshwater fish, plants, invertebrates and habitats cannot cope with, and/or compete with, some, predominately exotic, undesirable species that have been introduced to New Zealand and continue to expand into new areas (Townsend and Crowl 1991; Townsend 1996; Allibone 1999; Department of Conservation 2003; McDowall 2006a; McIntosh et al. 2010; Jones and Closs 2015). In these situations, exclusion barriers, which impede or prevent the upstream and/or downstream movement of undesirable fish species, can help protect key species populations and locations by keeping undesirable species out and providing a safe refuge area (Jones et al. 2021). Globally, exclusion barriers are increasingly used as a management strategy to control the spread of undesirable species that are a principal threat to freshwater biodiversity (Lintermans 2000; Lintermans and Raadik 2003; Tummers and Lucas 2019; Jones et al. 2021). In some locations non-migratory galaxiids have been inadvertently protected by man-made structures and natural waterfalls that prevent upstream migration of predatory fish. This management direction is supported by the NPS-FM, which identifies the need to prevent the passage of undesirable fish species to protect desirable fish species, life stages, or habitats (Section 2.3.2 for further details).

Exclusion barriers can also impact native species and habitats, and their effectiveness is situation-dependent. It is critical that an assessment of appropriate fish passage management for a waterway is made before installing, changing, or removing an instream structure. Consideration should be given to what species and habitats are present, their distribution and extent, their conservation status, habitat preferences, timing of migration and spawning, life history (e.g. Jones and Closs 2015), and possible impacts of providing or impeding fish passage (e.g., preventing or limiting range expansion of undesirable species, future fragmentation of a species, loss of genetic mixing, hybridising species, restricting some species from available habitats (Allibone 2000; Fausch et al. 2009; Woodford and McIntosh 2013), as well as the likelihood of success of management. Exclusion barriers do not only exclude fishes but also other species such as crustaceans, amphibians and invertebrates that will also need to be considered (Jones et al. 2021).

Exclusion barriers have successfully been used in New Zealand to protect native refuges and prevent access for undesirable species (Rowe and Dean-Spiers 2009; Department of Conservation 2012; Tabak 2020; Jones et al. 2021; Jack et al. 2023). They are generally designed to exceed the target fishes' ability to swim, jump or climb past the structure to manage their spread through the river network or into critical habitats. There are two main types of exclusion barriers, full barriers that prevent passage of all species upstream and downstream, and selective barriers that exclude undesirable fish while providing passage for desirable fish. A key motivation for the use of such barriers is that preventing invasion by undesirable species and reducing connectivity among habitats by barriers is generally a more efficient management strategy to control the spread of an undesirable species compared to trying to control population levels or eliminate a species after introduction (Vander Zanden and Olden 2008; Kates et al. 2012; Rahel 2013; Sherburne and Reinhardt 2016; Tummers and Lucas 2019). The following section focuses on the design of intentional exclusion barriers for undesirable fish species.

Exclusion barriers can be natural, physical, or behavioural. Natural barriers (e.g., waterfalls, chutes, dry reaches) are present throughout New Zealand and, depending on their location and parameters, they prevent access for some species and create native refuges upstream. Globally, physical barriers (e.g., weirs, exclusion screens, and velocity barriers) have been used most widely (47%), followed by electric (27%) and chemical barriers (12%) (Jones et al. 2021). Non-physical intentional barriers, such as acoustic, air, electric and light barriers, which stimulate an avoidance response, have been implemented internationally (Jones et al. 2021), with a small number tested in New Zealand. Experience to date suggests results have been mixed with overall low success (Bullen and Carlson 2003; Kates et al. 2012; Noatch and Suski 2012; Charters 2013; Ryder 2015; Jones et al. 2021). Generally, behavioural barriers can only be relied on when partial exclusion is acceptable and often need to be used in combination with intentional physical exclusion barriers to improve their effectiveness (Noatch and Suski 2012). At present, there is limited evidence available to provide national guidelines for the use of non-physical barriers in New Zealand and if behavioural barriers are considered they will need to be considered on a case-by-case basis and outcome monitoring undertaken.

Where fish passage is, or will be impeded, permits may be required from the Department of Conservation (DOC). Culverts and fords may not impede fish passage unless that impediment has been approved or exempted by DOC, and any dam or diversion may require a fish facility. In addition to these specific fish passage requirements, there are other approvals required for installing physical structures in streams (see Section 2 for further detail).

6.2 When must exclusion barriers be considered?

The undesirable fish species that are present or have the potential to invade a freshwater community needs to be considered when making any decision on appropriate fish passage management at a structure. Consideration should be given as to whether excluding the undesirable species will result in the protection or recovery of mahinga kai, threatened species and/or habitats, prevent new fish invasions, and if barriers will be viable in the prevailing environment. Iwi/hapū should be involved in determining what is a desirable or undesirable species (see Case Study 6 for an example).

Maintenance of known fish passage barriers (e.g., waterfalls, drying reaches or built structures found to be protecting key native refuges) should be considered when undesirable species could or are impacting on a location that supports key native fish populations and/or other biodiversity or cultural values/habitat. Generally, natural barriers to fish passage should not be removed or altered, unless conditions have changed, and undesirable species have gained access to a vulnerable habitat that is subsequently being negatively impacted. There are also some physical structures, such as culverts and dams, that should be retained as they have become fish passage barriers over time and valuable fish communities exist upstream of the barrier. In determining if instream structures should be maintained, enhanced, or removed, consideration should be given to the species that are currently found in these locations, what species should naturally be present, and whether maintenance or removal of the barrier is viable. Such decisions should be made in consultation with local iwi, the asset owner, DOC and the applicable council or territorial authority.

6.2.1 What fish species can be undesirable?

Of all the freshwater fish found in New Zealand, several introduced, and a few native fish species, have been found to impact on some freshwater species and key freshwater habitats in certain locations (McDowall and Allibone 1994; Department of Conservation 2003; McDowall 2006a; McIntosh et al. 2010).

Not all fish have the same impact on species and habitats where they have established (NIWA 2020), and risk assessments, known distributions, and evidence of impacts should be used to guide what fish species should be considered undesirable and where they are causing impacts within each waterway. For example, brown (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*), will be desirable and undesirable in different locations/situations due to their ability to co-exist with some freshwater communities and having significant impacts on other communities where they cannot co-exist (Allibone and McDowall 1997; McDowall 2006a; Jones and Closs 2015; Jack et al. 2023). For some undesirable species exclusion barriers could be considered in fish passage management decisions to aid their management in waterways.

At least 21 species of introduced freshwater fish have established self-sustaining populations in New Zealand waters (Department of Conservation 2003). Some of these species pose a threat to the health of native species through predation, competition, and/or changes to aquatic habitats (e.g., destabilisation of aquatic environments, loss of indigenous plant biodiversity, implications for health and cultural wellbeing, economic loss and reduced recreational activities) (Rowe and Dean-Spiers 2009; NIWA 2020). Assessments of ecological impact and overall risk by introduced fish species have been undertaken nationally (Department of Conservation 2003; Rowe and Wilding 2012; Collier and Grainger 2015; NIWA 2020), and these studies have predominately found the same introduced species as posing the greatest risk and threat to New Zealand's biodiversity: koi/amur carp[16] (*Cyprinus rubrofuscus*), gambusia (*Gambusia affinis*), perch (*Perca fluviatilis*), brown bullhead catfish (*Ameiurus nebulosus*), rudd (*Scardinius erythrophthalmus*), orfe (*Leuciscus idus*), brown trout, rainbow trout, tench (*Tinca tinca*), silver carp (*Hypophthalmichthys molitrix*) and goldfish (*Carassius auratus*) (Department of Conservation 2003; Rowe and Wilding 2012; Collier and Grainger 2015; NIWA 2020).

Species such as rudd, catfish, gambusia and orfe are known to compete for food with native species. Catfish, gambusia and perch are also known to directly predate native fish and macroinvertebrate species, and rudd are known to eat macrophytes, especially native species (Allibone and McIntosh 1999; Ludgate and Closs 2003; Rowe and Smith 2003; Collier and Grainger 2015; NIWA 2020). Koi/amur carp, rudd and catfish are known to disturb the ecology and freshwater communities that they invade (Collier and Grainger 2015; NIWA 2020). Some of these species have extended their distribution into new locations as a result of being manually introduced into new waterways, often illegally, and/or for some species, such as koi/amur carp that prefer warmer water temperatures, expanding their range as new habitats become more favourable in changing climates (Collier and Grainger 2015). Not surprisingly, koi/amur carp, gambusia, and rudd are recognised as pest fish with legislative status, classified as 'unwanted organisms' and/or 'noxious' under legislation (Biosecurity Act 1993, Freshwater Fish Regulations 1983). However, it should be noted that rudd are a licenced sports fish in the Auckland/Waikato region.

High risk fish species also include some established self-sustaining recreationally valuable fisheries, such as brown trout, rainbow trout and chinook salmon (*Oncorhynchus tshawytscha*). These species will be desirable in most locations but can also be undesirable in particular locations and habitats where they have not been found previously, and/or where they are impacting on native species or habitats. Although these high-risk species could be undesirable nationally, regionally, or at just specific locations, consideration needs to be given to the risk posed by each species in relation to the knowledge of their present distribution, the way each species spreads, and effective control methods

[16] The cyprinid previously known as common or koi carp, *Cyprinus carpio*, was redesignated to amur carp, *Cyprinus rubrofuscus,* in 2023.

available for their management before any decisions can be made on whether it is undesirable and if exclusion barriers could be effective and beneficial.

Salmon and trout species are implicated in the decline of some native fish populations, mostly through resource competition, but in some locations due to predation (Lintermans 2000; McIntosh et al. 2010; Woodford and McIntosh 2013; Jones 2014; Jellyman et al. 2017; Jones and Closs 2018; Holmes et al. 2021; Coughlan 2022). Galaxiid species are often prey for salmonids across New Zealand (McDowall 2006a; Jones and Closs 2018; Coughlan 2022). Non-diadromous galaxiids and mudfish are the most vulnerable groups to trout predation (Coughlan 2022). Trout prey on all life stages of galaxiid species and compete for food and space (McDowall 2006a). Trout predation has caused local extinctions and impacts on many of non-migratory galaxiids in New Zealand and globally (Townsend 1996; Allibone and McDowall 1997; Allibone and McIntosh 1999; Allibone 1999; Lintermans 2000; Jackson et al. 2004; McIntosh et al. 2010; Jones and Closs 2018). Species deemed at least risk from trout predation included torrentfish, eels, bullies, smelt (*Retropinna retropinna*), flounder, and pouched lamprey (Coughlan 2022).

Trout colonisation is not static and there are increasing reports of trout and other species moving into areas they have not previously occupied (McDowall 2006a; McIntosh et al. 2010; Jack et al. 2023). One factor influencing the distribution of trout is climate change. Changing thermal regimes can create thermal barriers to salmonid species movement, encouraging fish to seek cooler upstream waterways (Jackson et al. 2004; Hesselschwerdt and Wantzen 2018), causing the long-term security of all galaxiid species to be of increasing conservation concern (McDowall 2003; McDowall 2006b). In recent times there have also been observations of lower numbers of galaxiids in areas where brook char (*Salvelinus fontinalis*) have become established (Allibone and McDowall 1997; McDowall 2006a).

The impact of all introduced fish species should be considered, along with any local knowledge and experiences, to determine if they could be considered undesirable. Several introduced fish species have been recognised as having relatively high ecological risk, such as salmon, brook char, and mackinaw (*Salvelinus namaycush*), but impacts are not currently well known to allow better assessment (Department of Conservation 2003; Rowe and Wilding 2012; Collier and Grainger 2015). Fish and Game, Department of Conservation, Councils and mana whenua will have local and specialist knowledge and should be part of a team making decisions on what species could be considered undesirable, and if exclusion barriers could be helpful in their management or the protection of key native values (Holmes et al. 2021; Tadaki et al. 2022). Any previously eradicated fish species, such as gudgeon (*Gobio gobio*), or exotic species new to New Zealand should be given a high priority and considered for exclusion barriers, depending on the habitat they have invaded and the anticipated likely impacts (NIWA 2020).

Longfin eel, shortfin eel, Australian longfin eel (*Anguilla reinhardtii*), kōaro and non-migratory galaxiids (considered as a general group rather than specific species) were identified in Department of Conservation (2003) as the native fish species with the potential to cause negative impacts, though none were assessed as high risk compared to the introduced species. These native species are generally desirable species but can be undesirable in specific locations when they move into new freshwater communities and predate and/or outcompete established species (Case Study 5). Both kōaro and eels have been found to impact directly on other native fish populations, predominately non-migratory galaxiids and Canterbury mudfish (McDowall and Allibone 1994; Allibone and McDowall 1997; Allibone 2000; O'Brien and Dunn 2007; Tabak 2020).

Risks from native fish species arise primarily from when they are manually transferred to areas they would not naturally inhabit, or they increase in distribution or abundance in contrast to their historical range. This has occurred where land-locked populations of kōaro have become established upstream of large dams, resulting in the proliferation of this native species in areas of the catchment where they would not normally exist in such high abundance, or in habitats occupied by other native fish communities that they have not previously co-existed with (Allibone 1999; Tabak 2020).

Case Study 5: Ngāti Tahu Ngāti Whaoa – Kōura Restoration Plan

The vision of Ngāti Tahu-Ngāti Whaoa for mahinga kai is "To be able to provide healthy and plentiful mahinga kai for the Ngāti Tahu-Ngāti Whaoa people, visitors and for cultural events, tangi and other important occasions" (Ngati Tahu-Ngati Whaoa 2021). This is part of their cultural heritage and the ability of their waterways to "sustain and provide for Ngāti Tahu-Ngāti Whaoa people is integral to the iwi's wellbeing" (Ngati Tahu-Ngati Whaoa 2021).

The construction and operation of the Waikato River hydropower scheme has had substantial impacts on the composition of aquatic ecosystems and, hence, fisheries and mahinga kai in the Ngāti Tahu-Ngāti Whaoa rohe. Kōura (*Paranephrops planifrons*) are an important species for the iwi; forming part of their cultural identity, as a source of kai, as an indicator of waterway health, and as a means of expressing and practicing their cultural identity (Ngati Tahu-Ngati Whaoa 2021). While seeding of the Waikato hydro-reservoirs (and incidentally of the tributaries) with elvers from Karāpiro dam has benefited the tuna fishery, Ngāti Tahu-Ngāti Whaoa have raised concerns about the loss of kōura from their rohe. Tuna are known to eat kōura and Ngāti Tahu-Ngāti Whaoa have concerns about "the combined effects of high tuna numbers coupled with other pest fish such as brown bullhead catfish, rudd and gambusia on kōura" (Ngati Tahu-Ngati Whaoa 2021).

In response to concerns about declining populations of kōura in their rohe, in 2021 Ngāti Tahu Ngāti Whaoa produced Te Haerenga Whakaoranga Kōura, a Kōura Restoration Plan. The plan states that "Kōura are now considered by some of our whānau to be a greater delicacy than tuna" (Ngati Tahu-Ngati Whaoa 2021) and sets out goals and actions to reduce the impacts of predators on kōura by preventing tuna, trout and pest fish from accessing kōura populations above natural and artificial fish passage barriers. It has been identified that "Sometimes having barriers to fish movement might be beneficial for kōura by reducing predator access. If there is a barrier such as a natural drop or even a perched road culvert downstream of your site, then it might be beneficial to leave the barrier in place. It might already be helping stop other fish species accessing your site and provide you with an easier task in restoring kōura" (Ngati Tahu-Ngati Whaoa 2021).

In response to the plan, we understand that the special permit for the Karāpiro dam elver trap-and-transfer no longer allows elver releases to the two most upstream reservoirs (Lakes Ohakurī and Ātiamuri) in response to the concerns of Ngāti Tahu Ngāti Whaoa (Boubée et al. 2022).

Interspecific competition between non-migratory galaxiid species may also require intervention. For example, alpine galaxias (*Galaxias paucispondylus*) may compete for, and dominate, habitats with other smaller upland longjaw galaxias (*Galaxias prognathus*). Active management of alpine galaxias through exclusion barriers may be warranted if declines in longjaw galaxias are observed/confirmed. Also hybridising risk may need consideration for exclusion barrier management as some hybrid populations, which should be avoided, have resulted from one non-migratory galaxias being translocated via intake races transferring water across catchments into a new waterway where another non-migratory galaxiid existed (Allibone 2000).

If the objective is to aid in protection of key native species and habitats, then the fish species identified above are those that if present in some waterways could be considered undesirable to provide fish passage for (as required by NPS-FM; Section 2.3.2). Assessing if these fish species will impact a location, and whether excluding the undesirable species will result in the protection or recovery of threatened species and/or habitats, or prevent new fish invasions, will be required before a species is deemed undesirable.

6.2.2 Which desirable native fish may benefit from exclusion barriers?

Non-migratory species likely to benefit most from exclusion barriers

Of our native fish, non-migratory galaxiids and mudfish are the key taxa where populations could benefit most from a natural or full exclusion barrier (see Table 6-1). These species often have small, isolated and fragmented distributions, occur in, or are restricted to, habitats that are conducive to exclusion barriers, do not require access to and from the sea to complete their lifecycles, can maintain a self-sustaining population upstream of barriers, and are vulnerable to direct predation and/or competition by undesirable species and to the adverse changes to aquatic habitats caused by some undesirable species (Rowe and Dean-Spiers 2009; Salant et al. 2012). The majority of non-migratory galaxiids are nationally threatened or at risk (Dunn et al. 2018) and are impacted by a range of factors including undesirable fish. If the current rate of documented losses continues, then for some of these species, we may see extinctions within the next century (Bowie et al. 2013; Dunn et al. 2018).

Salmonids have established self-maintaining populations in many waters not targeted or managed as fisheries. These are often the habitats of non-migratory galaxiids and where there is a need to balance the necessity to protect an increasingly threatened native fish fauna and the requirement for a highly valued trout fishery (Jellyman et al. 2017; Holmes et al. 2021). Jones (2014) found that differences in non-migratory galaxiid life histories had implications for how they can interact with undesirable salmonids. He found those species that have a fast life history with small eggs, high fecundity, and are predominately found in highly productive, frequently disturbed, low to mid catchment waterways can co-occur with salmonids (e.g., Canterbury galaxias, Central Otago roundhead galaxias) (Jones 2014; Jones and Closs 2015; Jones and Closs 2018). The large numbers of dispersive larvae in these lower catchment species support a source-sink metapopulation system (Woodford and McIntosh 2013) whereby populations in salmonid-invaded reaches are sustained by immigration from upstream salmonid-free refugia (Jones and Closs 2015). In contrast, those species with a slow life history, found in low productivity, but stable, headwater waterways with larger egg size and lower fecundity were often impacted and excluded by salmonids (e.g., dusky galaxias, Eldon's galaxias) (Jones 2014; Jones and Closs 2015; Jones and Closs 2018). Low larval abundance and poor dispersal in these headwaters mean they form isolated populations, so if salmonids invade there is limited dispersal from upstream to counter impacts and extinction is likely (Jones and Closs 2015). Some species such as the Taieri flathead galaxias (*Galaxias depressiceps*) and Southern flathead galaxias (*Galaxias 'southern'*) were found to have intermediate life histories and would sometimes be impacted by salmonids (Jones 2014; Jones and Closs 2015; Jones and Closs 2018). These differences in life histories have been considered when identifying the priority desirable species that could benefit from the security of an exclusion barrier (Table 6-1). For some of these species, such as lowland longjaw galaxias, dusky galaxias, Eldon's galaxias, Clutha flathead galaxias, Nevis galaxias and Teviot flathead galaxias, it has been found that without natural waterfall barriers and/or exclusion barriers, or conservation management intervention (including undesirable species removal and barrier installation to prevent reinvasion), these populations would have or have been lost, and could now be extinct (Allibone and McDowall 1997; Department of Conservation 2003; Bowie et al. 2013; Jones 2014; Jones and Closs 2018).

Table 6-1: List of key non-migratory galaxiids that could have increased protection from a natural or exclusion barrier to exclude undesirable fish.

Common Name	Scientific Name	Built or natural barriers would be advantageous to prevent extinction (High (H), Medium (M))
Central Otago roundhead galaxias	*G. anomalus*	M
Lowland longjaw galaxias	*G. cobitinis** except for Kauru and Kakanui	H
Taieri Flathead galaxias	*G. depressiceps*	M
Dwarf galaxias	*G. divergens*	M
Eldon's galaxias	*G. eldoni*	H
Gollum galaxias	*G. gollumoides*	M
Bignose galaxias	*G. macronasus*	M
Alpine galaxias	*G. paucispondylus*	M
Upland longjaw galaxias	*G. prognathus*	M
Dusky galaxias	*G. pullus*	H
Clutha flathead galaxias	*G. 'species D'*	H
Northern flathead galaxias	*G. 'northern'*	M
Canterbury galaxias	*G. vulgaris*	M
Dune lake galaxias	*G. 'dune lakes'*	M
Southern flathead galaxias	*G. 'Southern'*	M
Teviot flathead galaxias	*G. 'Teviot'*	H
Nevis galaxias	*G. 'Nevis'.*	H
Canterbury mudfish	*N. burrowsius*	M
Brown mudfish	*N. apoda*	M
Black mudfish	*N. diversus*	M
Northland mudfish	*N. heleios*	M

Other non-migratory native fish species could also benefit from full or selective barriers, but it is unlikely that exclusion barrier management is required for these species to be effectively protected, and they should be considered on a case-by-case basis.

Other species that could benefit from exclusion barriers

In addition to non-migratory galaxiids and mudfish, other native fish like kōaro, upland bully (*Gobiomorphus breviceps*), giant kōkopu, shortjaw kōkopu, Tarndale bully (*Gobiomorphus alpinus*) and bluegill bully (*Gobiomorphus hubbsi*) are thought to be the next most vulnerable to trout and could benefit from exclusion barriers (Coughlan 2022).

Migratory native fish species can benefit from a selective barrier that provides access for climbing species over a natural or exclusion barrier, (e.g., banded kōkopu and giant kōkopu), while preventing other non-climbing undesirable species (e.g., trout, perch, koi/amur carp) from moving upstream. For example, waterfalls can maintain a good native fish refuge from introduced species. By preventing undesirable fish access, these selective barriers provide access for young native fish to protected upstream habitats including spawning habitats for adult fish. Whether migratory species, such as the large galaxiids and smelt (McDowall 2000), can develop facultative non-migratory lifecycles, which allow them to maintain a self-sustaining population if they were to be isolated, is another key consideration in determining if species may benefit from a selective barrier. However, careful consideration needs to be given to the trade-off between effectively fragmenting populations, isolating catchments areas to reduce the spread of undesirable species, and the need to maintain river connectivity for desirable species (Porto et al. 1999; Peterson et al. 2008; Fausch et al. 2009; Jones et al. 2021).

6.3 Factors to consider in creating and maintaining an exclusion barrier

6.3.1 Key considerations when setting objectives for exclusion barriers (selective or full)

Exclusion barriers are maintained or installed with the objectives of preventing or reducing undesirable species by impeding undesirable species passage and protecting the desirable species and/or habitat. It is crucial that objectives and performance measures are set when deciding if or where an exclusion barrier is possible and as part of finalising the design (See Section 3 for guidelines). In some locations, non-migratory galaxiids have been inadvertently protected by man-made structures that prevent upstream migration of predatory fish; it is important to also consider site-specific information prior to confirming any future structure objectives before works are undertaken.

For successful exclusion barriers, biological traits and behaviours of the undesirable and desirable species should be considered (Section 6.3.2), along with hydrological and physical features (Section 6.3.7), to create a design that exceeds the undesirable species' abilities e.g., jumping, swimming, or climbing ability (Noatch and Suski 2012; Tummers and Lucas 2019). Biological, hydrological, and physical performance measures will be important and may be opposite to those for the remediation of fish passage for desirable species (Section 3).

In addition to setting general objectives and performance standards, specific consideration will need to be given to the stage of invasion of the undesirable species, the species being excluded, whether a selective or full exclusion barrier is appropriate, and the impacts of exclusion (Figure 6-1). It is important for decision-makers to consider any trade-offs between set objectives and performance measures that will enable successful design, installation, and maintenance of the exclusion barrier. The following questions should be considered when setting objectives for exclusion barriers.

Stage of invasion?

Exclusion barriers may also be considered to simply prevent further spread, reduce pressure, prevent spawning and protect particular habitats, or prevent access for undesirable species, to aid recovery of important freshwater habitats that they have impacted (Tempero et al. 2019), rather than solely to protect an individual species or population. It should be noted that intentional screened barriers (e.g., water intakes) are the exception, and should generally exclude all species (Hickford et al. 2023), as otherwise these fish are lost to the fishery, especially diadromous species.

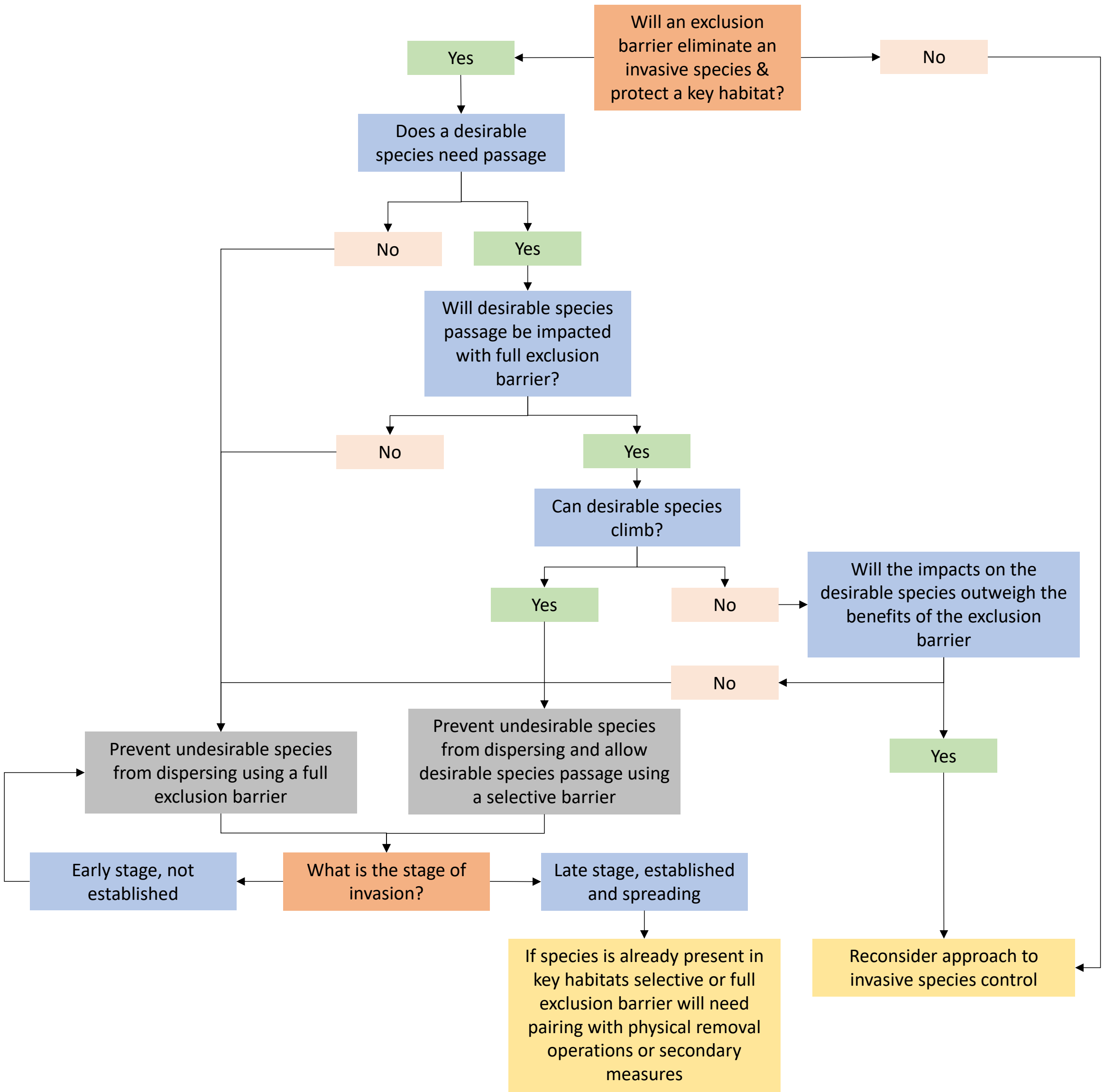


Figure 6-1: Setting objectives and determining the type of barrier applicable for control of the undesirable species.

Full or selective exclusion barrier?

Whether to install a full or selective exclusion barrier will depend on the situation, life history and ecology requirements of the species present and/or the habitat being protected from undesirable species.

If diadromous species' strongholds are present and are proposed to be protected from an undesirable species, a selective barrier will likely be required to ensure the diadromous species can negotiate the barrier to maintain their migratory life cycle or ensure life stages can still migrate or disperse. There could be exceptions to this in limited situations, including where diadromous species have formed landlocked populations and can complete their lifecycle within the barrier area. Where barriers will create a new lacustrine population, consideration should be given to the potential impacts on other native species through altering the distribution and abundance of competing species outside of their normal range.

In New Zealand, full exclusion barriers will predominately be required where highly-threatened non-migratory galaxiids are restricted to fragmented headwater locations, and where without a barrier these populations are likely to become extinct. Fortunately, the distance inland to these headwater sites means diadromous fish species are effectively absent from many non-migratory galaxiid sites and, therefore, passage past the barrier for diadromous species is not required. Once an initial barrier has been installed, additional barriers and invasive species removal operations can be established over time and expand further downstream to extend the range and protected area for the non-migratory galaxiid species (Lintermans 2000).

Internationally, selective barriers have been difficult to achieve (Jones et al. 2021). However, in New Zealand, we have had some good success as often the undesirable and desirable fish species have different behavioural traits that can be exploited. Figure 6-1 outlines key considerations in deciding if a full or selective barrier is appropriate for the species and site.

How will desirable species be impacted by the exclusion barrier?

In each situation, consideration needs to be given to the possible impacts of providing or impeding fish passage (Figure 6-1). This includes restricting some species from reaching available natural habitats, potential fragmentation of a species, the possibility of creating sink populations, isolating populations, risk of localised extinction, ensuring adequate habitat quantity and quality for sustaining populations, the loss or restriction of the ability to carry out full lifecycles within the barrier area, and loss of genetic mixing that could affect the long-term resilience of the species (Allibone 2000; Eikaas and McIntosh 2006; Fausch et al. 2009; Woodford and McIntosh 2013; Jones et al. 2021). Knowledge of source and sink population dynamics will be important when making decisions on where exclusion barriers should be established and ensuring that these barriers do not disrupt dispersal ability (Woodford and McIntosh 2010).

Where exclusion barriers prevent dispersal of native species, the size of isolated populations needs to be considered very carefully or populations will experience gradual decline (Muhlfeld et al. 2012). Minimum viable population sizes have rarely been considered when implementing exclusion barriers, which have often been used as a last-ditch effort to protect populations from extinction. Further research is required to establish minimum viable population sizes for a greater range of native species to better inform the long-term effectiveness of exclusion barriers (Jones et al. 2021).

6.3.2 Biological traits, behavioural, and life history considerations

Different freshwater fish have different traits, abilities, and characteristics (e.g., physical, physiological, sensory; Appendix A). These differences, including swimming ability, can be exploited to identify key design parameters to limit or prevent undesirable species' movements over or through a structure, while other features can allow some desirable species to navigate the structure, depending on if the purpose is to create a full or selective exclusion barrier (Table 6-2). Most exclusion barriers currently relate to limiting upstream movement. However, the same principles can also be applied to lateral connections and downstream movement, although downstream movement, especially 'drift' dispersal and colonisation, can make it more difficult (Tummers and Lucas 2019).

Table 6-2: Factors influencing fishes' ability and likelihood of successfully negotiating barrier(s). Adapted from Charters (2013) with consideration of water temperature and hydraulic wave added from Holthe et al. (2005) and Stuart (1962) respectively. Originally adapted from Rowe and Dean-Spiers (2009) and Noatch and Suski (2012).

Fishes Ability/Response	Influencing Factors
Jumping	Height of barrier Longitudinal distance from downstream pool to top of barrier Area of downstream pool Depth of downstream pool Fish species Age and size of fish (i.e., juvenile versus adult) Water temperature
Upstream swimming	Fish species Age and size of fish (i.e., juvenile versus adult, certain fish will be excluded by screen gap size) Water velocity/hydraulic wave High flow conditions (i.e., floods) Maximum swimming speed of fish Water depth in stream channel (e.g., juvenile fish can move upstream in less water than adult of same species)
Climbing	Fish species Availability of wetted surface (for adhesion)
Avoidance response	Sensitivity range of fish species to electricity or environmental conditions such as sound, light, and water pollutants

Different exclusion barrier designs will be needed to prevent different undesirable species (Section 6.2.1). For example, trout can jump, while some native fish can climb, so in these situations fall height, grates or overhangs, or a lack of water depth downstream can be used to prevent trout jumping upstream, while still allowing climbing native species access. Some fish may be excluded by providing high water velocities or other flow conditions that undesirable species cannot negotiate (Appendix B). If climbing fish, like eels and kōaro, need to be prevented from moving upstream, then an overhanging lip that prevents climbing will be required.

In addition to the fish's abilities, their behaviour should also be considered, as not all species will be affected equally by a barrier. For example, an eel may be able to navigate around an instream structure via land in a way that cannot be achieved by whitebait or trout. Avoidance responses of fish may also be exploited, including the use of electricity, or species' responses to environmental conditions such as sound, light, and water pollutants (Benejam et al. 2015; Johnson et al. 2016). However, non-physical barriers can only be relied on when selective exclusion is acceptable and often need to be used in combination with intentional physical exclusion barriers to improve their effectiveness (Noatch and Suski 2012).

Exclusion barriers can generally be categorised into 'high head' (>1.0 m) or 'low head' (<1.0 m) barriers. Achieving head drops of >1 m requires relatively steep stream gradients if significant impacts on upstream habitats (e.g., backwatering) are to be avoided. Most exclusion barriers are needed to prevent upstream movement, as the undesirable species of concern establishes in the lower reaches, and for many at risk species, headwater areas are the only remaining strongholds. However, at times there could be a need to control downstream movement. For instance, where an undesirable species has been introduced into a lake environment, a barrier to prevent the undesirable species establishing in a downstream location may be desired, or a barrier may be established to collect undesirable species as they move downstream and prevent any upstream passage back into an area where restoration is being attempted, or barriers may be installed to control/decrease adults getting into key habitats (e.g., Figure 6-8 & Figure 6-10; Section 6.3.4 below).

6.3.3 Salmonid exclusion barriers considerations

Trout and other salmonids negotiate structures by jumping, and often utilise plunge pools to leap and navigate past barriers. As detailed in Appendix B, brown trout and other salmonids, such as brook char, have good jumping abilities compared to our native fish, with some able to jump greater than 0.74 m fall height, depending on fish size and condition, and surrounding waterway conditions (e.g., downstream pool size and depth) (Aeserude and Orsborn 1985; Holthe et al. 2005; Kondratieff and Myrick 2006). The inability of salmonids to negotiate >1 m high vertical barriers makes high-head exclusion barriers ideal for excluding these species (Figure 6-2; Figure 6-3; Figure 6-4; Figure 6-8). In addition to barrier height, from installations to date it has been found that a ≥0.5 m overhang along with height >1 m is thought ideal to inhibit jumping (e.g., Figure 6-5, Case Study 7). Tabak (2020) found perched pipe culvert designs could be used as migration barriers to limit these undesirable species. Exclusion barriers designed to prevent salmonid access should, therefore, focus on ensuring the structure has a fall height significantly greater than 0.74 m and/or have a ≥500 mm overhang to prevent access. In addition to these features, full exclusion high head barriers in Australia have had success in preventing undesirable fish access by infilling the upstream pool and by placing large boulders on the margins of the barrier that ensure any flood flows are directed to the middle of the channel to prevent breaches around the structure (Figure 6-8).

The importance of barrier height is highlighted by some natural waterfalls in New Zealand and Australia that were previously protecting key non-migratory galaxiid strongholds but have changed over time. These natural structures needed augmenting with the installation of barriers on top of the waterfalls to prevent undesirable species access (Figure 6-2 & Figure 6-3) (Sanger and Fulton 1991; Jack et al. 2023).

If partial exclusion is required in these situations, then the differing behaviours of the desirable and undesirable fish may be exploited, e.g., climbing native fish compared to jumping salmonids, or selective fish passes can be considered (Case Study 6).

In situations where fall heights of >1 m cannot be achieved, low head exclusion barrier designs will have to strengthen other design features that will prevent access, e.g., providing a shallow water zone downstream to prevent salmonids jumping and adding bars or screens that will prevent access upstream (see Table 6-3; Figure 6-5; Figure 6-6; Figure 6-7) (Lintermans and Raadik 2003). One-way barriers (Figure 6-8 & Figure 6-10) (Tempero et al. 2019), good water intake screen designs (Hickford et al. 2023), or non-physical barriers, e.g., electric (Noatch and Suski 2012), could also be considered in these situations, but outcome monitoring is critical to ensure success.

Figure 6-2: Swinburn Creek barrier in Otago. A high head exclusion barrier has been installed onto a natural waterfall to protect the Central Otago roundhead galaxias population, after brown trout gained access upstream when stream conditions changed. Key features of the barrier are the height to prevent trout access. Photo credit: Daniel Jack.

Figure 6-3: Akatore Creek barrier in Otago. A high head exclusion barrier installed onto a natural waterfall to protect Taieri flathead galaxias, after brown trout gained access upstream when conditions changed at the natural waterfall. Key features of the barrier are the height to prevent trout access, the overflow to minimise backwater creation and loss of non-migratory galaxiid habitat, and the wood boards to manage height and enable flushing if required. Photo credit: Sjaan Bowie.

Figure 6-4: High head exclusion barrier installed in Fork Stream, Canterbury. The barrier was built to prevent trout accessing a key non-migratory galaxias stronghold in Fork Stream, Upper Waitaki River, Canterbury. Key features of the barrier were the height and downstream shallow zone (concrete splash pad) to prevent trout access, and the adjustable boards that can be slotted in and out to flush the habitat, minimise backwater and increase height if needed. Photo credit: Sjaan Bowie.

Figure 6-5: Low-head exclusion barrier installed in an unnamed spring, Waterwheel Wetland, Canterbury. The objective was to prevent trout and kōaro accessing a key bignose galaxias stronghold in the MacKenzie Basin, Canterbury. Key features of the barrier were the height (0.31 m fall from end of pipe to concrete splash pad) and downstream shallow zone (1.95 m of concrete splash pad below the pipes) to prevent trout access and the 0.62 m height and dry concrete face of the wall above the concrete pad to prevent kōaro and trout passage. Photo credit: Sjaan Bowie.

Figure 6-6: Exclusion barrier installed in an unnamed spring of the Fraser River. The objective was to prevent trout and kōaro accessing a key lowland longjaw galaxias and bignose galaxias stronghold in the MacKenzie Basin, Canterbury. Key features of the barrier were the height (0.83 m) and shallow downstream zone (concrete splash pad) to prevent trout access, the metal lip (300 mm wide at barrier face, 100 mm on concrete sides, 150 mm on wooden sides of bridge) to prevent kōaro passage (seems to be unsuccessful), and the wooden drop logs to allow for flushing of flows and maintenance of flows and habitat upstream. Photo credit: Sjaan Bowie (DOC).

Figure 6-7: An example of an anti-jump screen/ grill that could be added to a barrier to prevent trout access. Photo credit: Sjaan Bowie.

Figure 6-8: Example of a full exclusion barrier protecting a non-migratory galaxiid, Shaw galaxias, in Australia. The objective was to prevent trout accessing a key galaxias in Australia. Key features of the barrier were the height (1.3 m), shallow downstream zone to prevent trout access, infilling the upstream pool to provide stability for the barrier and prevent habitat for trout to jump into and placing large boulders on floodplain to direct overflow into the middle of the channel. Photo credit: Tarmo Raadik.

Case Study 6: Ngāti Rangiwewehi – Exclusion barriers to support kōaro populations

Kōaro were once the dominant fish species in most of the large, inland lakes of the central North Island. Kōaro populations in the Te Arawa Lakes were decimated by the introduction of trout in the late 1800s and further reduced following introduction of smelt to Lake Rotorua in the 1920s, and to the other Te Arawa Lakes in the 1930s (Rowe and Kusabs 2007). The introduction of exotic fish to the Te Arawa Lakes is a significant threat to customary fisheries, including for kōaro.

The decline of kōaro in the Ngāti Rangiwewehi rohe prompted the establishment of the Kōaro Restoration Project. The project was a collaboration involving Bay of Plenty Regional Council, Te Arawa Lakes Trust, Ngāti Rangiwewehi, DOC, and Fish and Game Eastern Region. The aim of the project was to trial options for trout removal from kōaro habitat and to evaluate the population effects of these interventions on kōaro. The hope was to provide a template for kōaro restoration that could be applied more widely across Te Arawa Lakes catchments, fulfilling key objectives in Mahire Whakahaere, the Te Arawa Lakes Fisheries Management Plan (Te Arawa Lakes Trust Te Komiti Whakahaere 2015).

An opportunity was identified in Hamurana Springs, a tributary of Lake Rotorua, to establish an exclusion barrier that would prevent trout from accessing the stream and protect the resident kōaro population. A selective barrier was designed and installed in Hamurana Springs in 2012. The structure took the form of a concrete weir across the stream that juvenile kōaro could climb to access the stream. Trout were excluded through the addition of an anti-jump screen and a concrete apron that prevented trout from being able to jump over the weir. Following installation of the weir, any trout remaining upstream were relocated downstream of the weir.

Subsequent monitoring initially indicated increases in kōaro numbers in the protected habitat upstream of the exclusion barrier. However, the backwatering effect caused by the weir has negatively impacted habitat quality in the reach immediately upstream of the weir and the weir has had to undergo several modifications following the detection of trout incursions above the weir. This highlights the difficulties in effectively implementing selective exclusion barriers, particularly in low gradient streams.

Figure 6-9: Trout barrier at Hamurana Springs. Photo credit: Andy Woolhouse.

6.3.4 Koi/amur carp exclusion barrier considerations

When confronted by barriers, carp have been found to jump over or push through them (Thwaites et al. 2010). Carp also have differences in morphology, predominately being wider in body, compared to most desirable fish and these features can contribute to barrier design, where large carp can be excluded, and smaller native fish can pass. Carp also undertake seasonal migrations to spawn, and spawn in the same locations each year, so barriers could use this trait to limit spawning movements (Taylor et al. 2012; Piczak et al. 2023). Their biological traits, behaviours, and differences in reproductive and migration timing, can be considered in barrier design to help exclude carp (Piczak et al. 2023).

Sensory sensitivity could be exploited in situations where carp display sensitivities to electrical, acoustic, visual and/or chemical stimuli, however, studies to date have found variable success (Piczak et al. 2023). Some international studies have shown carp will avoid electrical (Johnson et al. 2016; Kim and Mandrak 2017a; Bajer et al. 2018; Piczak et al. 2023), strobe, acoustic (Kim and Mandrak 2017b; Bzonek et al. 2021) and bubble curtain stimuli (Zielinski et al. 2014; Zielinski and Sorensen 2015) but responses can vary with season, life stage and environmental variables. Consequently, researchers are now integrating sensory stimuli to produce multi-modal cues that take advantage of all cues collectively. For example, ensonified bubble curtains, a stimulus where sound is projected into an air bubble stream to focus and enhance sound fields, while resonating the air bubbles (Dennis et al. 2019; Feely and Sorensen 2023). The Bioacoustic Fish Fence (BAFF) (https://www.fgs.world) a commercial ensonified bubble curtain, has been tested in the laboratory and field with >92% success at deterring four carp species (Dennis et al. 2019; Martin et al. 2021; Feely and Sorensen 2023). Although these multi-sensory behavioural barriers have potential to exclude carp species, the responses of native New Zealand species have not been tested. Currently, there is insufficient evidence available to provide national guidelines on best practice for the use of non-physical barriers in excluding carp within New Zealand.

Physical barriers, such as weirs, culverts, active (e.g., trap and sort) or selective barriers (e.g., one way gates), have been used to prevent carp access, or to trap them (Jones et al. 2021). Eradication of carp is difficult and previous management efforts have used barriers to contain carp and decrease or eliminate access to specific habitats (Piczak et al. 2023). There has been varying success in the use of physical barriers to exclude carp overseas and in New Zealand (Lougheed et al. 2004; Hillyard 2011; Tempero et al. 2019; Tummers and Lucas 2019). The successful cases of koi/amur carp exclusion in Australia have used exclusion screens and traps that exploited biological differences between species, and utilised seasonal habitat use by carp (Hillyard 2011; Taylor et al. 2012; Stuart and Conallin 2018; Piczak et al. 2023). Most barriers designed for carp exclusion are selective barriers. Consequently, desirable fish and undesirable juvenile fish can continue to pass.

In Australia, they have found success in effectively filtering out carp from desirable species into traps placed at dams and weirs by exploiting carp's unique jumping behaviour with a cage device known as a Williams' Cage (Stuart and Conallin 2018) (Figure 6-10). The Williams' cage is a structure made up of vertical bars (with 4.2 cm spacing) that first captures desirable fish and common carp in a cage, then the desirable fish pass through via a false floor while the carp are captured in a second cage by jumping a low 30 cm barrier that sits above the water (Figure 6-10). The carp are then trapped in a holding cage and removed manually. Application of the Williams' cage could be effective in New Zealand where desirable species do not jump. However, it requires ongoing maintenance and clearing which can be labour intensive over time.

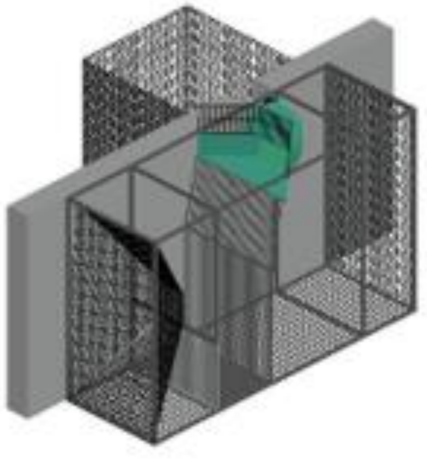


Figure 6-10: Example of a Williams Cage. The objective is to capture carp and desirable species, then use a low barrier to attract the carp into a trap. Key features of the barrier include vertical bar screens with 4.2 cm spacing, a false floor for desirable passage and a low 30 cm barrier for carp to jump. Source: Piczak et al. (2023).

Exploiting carp's pushing behaviour, one-way 'finger' (spaced 3.1 cm apart) structures are designed to be pushed through by adult carp and entrap them in a holding cage and effectively separate adult carp from desirable species (Conallin et al. 2016). However, further studies are still needed to assess desirable species' pushing ability before these barriers can be used (Piczak et al. 2023). Like Williams' Cage barriers, active removal of carp is required for these exclusion barriers to work.

Mesh screens with spacing ranging 0.3–5.0 cm have shown some success internationally at preventing carp access (Hillyard 2011; Piczak et al. 2023), but to date there has been no New Zealand applications, other than as part of overall water intake design to prevent impingement and entrainment generally (Hickford et al. 2023). Mesh screens and the design guidelines in Hickford et al. (2023) could be considered if full exclusion was sought and inundation with low velocity waters outside the screened area didn't occur during flood flows.

Internationally, vertical bar screens with spacing ranging from 3.1–9.0 cm have shown success in preventing movement of large-bodied carp (French et al. 1999; Lougheed et al. 2004; Hillyard 2011). Weaknesses of these morphological barriers (mesh and bar screens) include clogging and inhibiting the navigation ability of desirable fish and so need careful consideration (Piczak et al. 2023). Understanding the morphology of desirable and undesirable species is crucial for vertical bar screen designs, giving due consideration to the trade-off between what size undesirable fish needs to be excluded (exclusion of sexually mature fish is crucial as a minimum), but which also allows desirable species access when aiming for successful control, exclusion, or eradication. Often not all undesirable species can be excluded with vertical bar screens if access for desirable species is needed, so they are considered selective exclusion barriers.

In New Zealand, the installation of a one-way barrier, with 30 mm bar spacing, had limited success in preventing adult koi/amur carp access to a small lake (Tempero et al. 2019). The screened gate (Figure 6-9 and Figure 6-10) was designed to prevent adult carp moving upstream into Lake Ohinewai, Waikato, while allowing juvenile species to move upstream, and all desirable fish that are smaller to move into and out of the lake.

Although juvenile carp were able to pass upstream, it was deemed impractical to design a barrier capable of blocking all invasives, considering the conflicting need to provide free passage for native species. Tempero et al. (2019) found adult carp were excluded and the estimated carp population was reduced to low abundances in the short to medium term in the lake because of the barrier and active removal. However, it was determined that continued removal would be required to keep the carp population below levels that are likely to result in environmental impacts long term, unless non-migratory resident carp were eradicated, and smaller carp were prevented access. In addition, the reduction in the koi/amur population resulted in catfish, another undesirable benthivore, to increase substantially in abundance. This highlights the need for eradication or control programmes to consider the community of invasive fish species within a waterbody rather than having a single species focus. Some issues were also found with debris so future installations need to consider vertical bar placement that improves debris clearance (John Gumbley, pers. com). Tempero et al. (2019) suggested consideration could also be given to reducing the spacing of the vertical swing bars at the bottom of the trap or using a finger trap style one-way door (Thwaites et al. 2010) in any future installations, as it may improve the effectiveness of the barrier by blocking the passage of smaller fish. It is recognised that to be effective, these gate barriers will need to be used in combination with physical removal and outcome monitoring.

Figure 6-11: One-way gate fitted to Lake Ohinewai outlet drain, Waikato. The objective was to establish a selective exclusion barrier to prevent large koi/amur carp from migrating upstream into the lake, while allowing juvenile native species to move upstream, and all fish to move downstream to exit the lake. Photo credit: Adam Daniel.

Figure 6-12: One-way gate structure prior to installation showing screen that can be lifted for inspection and clearing of debris. Photo credit: Adam Daniel.

6.3.5 Undesirable native fish exclusion barrier considerations

At times, and in particular habitats, a few native fish, e.g., kōaro and eels, have moved into new habitats that they have not naturally been found in and may need to be excluded to protect desirable species that cannot compete with them (Allibone 1999; Bowie et al. 2010; Bowie et al. 2013). Kōaro and eels are both excellent climbers, especially when young, so exclusion barriers designed to exclude these species will need to account for this trait by adding perches and overhangs that cannot be navigated.

A variety of overhangs have been added to exclusion barriers in New Zealand with the aim of excluding kōaro (Figure 6-5; Figure 6-6; Figure 6-7). In laboratory trials, Tabak (2020) found juvenile kōaro were the best climbers and that longer fish with less body weight were more successful climbers than heavier fish of similar length. McDowall (2003) found the climbing behaviour and morphology of kōaro allows them to use wetted margins and either smooth or rough surfaces to navigate instream obstacles. Consequently, to prevent navigation, any wetted margin forming on the structure needs to be broken to prevent kōaro passage (Tabak 2020).

There have been many failed attempts to create a kōaro barrier (e.g., Figure 6-6), however, a >0.5 m perched culvert (Figure 6-5) and a solid downstream plate (Figure 6-13) added to high head weirs have successfully excluded kōaro. The successful solid plate added to the weir to exclude kōaro was the third attempt to design a kōaro barrier at this site (Tabak 2020). This lip has several design aspects that have aided its success including:

- Being made of aluminium to resist corrosion, and it is cheaper, easy to work with, light and transportable while still being strong.
- The perched lip was designed to:

- Be as wide as possible to prevent any water tracking along the lip and any wetted margin forming under any flow conditions. Topside panels on either end of the lip were added as an additional feature to stop water if it did track along the top during rain or splashing events.
- Be deep enough to keep a strip of concrete beneath it dry (either side of the main flow) under all flow conditions, but not too deep that water and any debris coming downstream could damage the barrier or get stuck.
- Be placed as high above the downstream water level as possible to stop algae or other things bridging the barrier and to stop any possible jumping opportunities.
- Be angled 120-degree downward with an additional thin downward facing lip with a drastic change in angle to discourage kōaro climbing.
- Have support struts on the topside of the barrier to keep the underside as flat as possible, and not create wetted climbing surfaces underneath.
- Ensure a tight seal against the concrete and stop water passing behind the barrier.

Figure 6-13: Full exclusion barrier installed in an un-named tributary of Upper Waipori River. The objective was to prevent trout and kōaro access to a key dusky galaxias stronghold in Otago. Inset shows a close-up view of the successful kōaro lip barrier. Key features of the barrier include the 175 mm depth aluminium sheet with 100 mm angle, sealed to the weir face with no gaps at a 120 degree downwards angle. Photo credit: Josh Tabak.

6.3.6 Consideration of providing selective passage for desirable native species

Vertical barriers have been designed that can successfully allow passage of desirable fish by exploiting the inferior leaping and climbing behaviour of undesirable species. There are a few successful New Zealand examples of selective exclusion barriers preventing access of undesirable fish while providing desirable migratory species passage (Figure 6-14 and Figure 6-15). These barriers use fall height >1 m and shallow downstream sections to prevent access for undesirable swimming species, while providing wetted margins and roughness for the passage of desirable species that all use climbing to migrate upstream. These exclusion barriers protect habitat for important lacustrine banded and giant kōkopu, as well as for longfin and shortfin eels, from the adverse impacts of koi/amur carp, goldfish, perch, rudd, gambusia and catfish.

Figure 6-14: A weir built on top of a natural waterfall in Waitawhara Stream, Waikato. Although not designed as a selective barrier, the weir and waterfall prevent koi/amur carp, goldfish, perch, rudd, gambusia and catfish from accessing headwater habitats while allowing the passage of banded and giant kōkopu and shortfin and longfin eels. Key features of the barrier include the roughness and wetted margins that desirable fish use to migrate, plus a rounded crest on the concrete weir. Photo credit: Cindy Baker.

Figure 6-15: Baffled concrete weir in the lower reaches of Puketirini Stream, Waikato. A degraded weir (inset top left) was successfully preventing koi/amur carp, goldfish, perch, rudd and gambusia from accessing upstream habitats while allowing the passage of banded and giant kōkopu, and shortfin and longfin eels. To continue protecting native fish populations, the weir was remediated in 2018. Key features of the barrier include the roughness and wetted margins that desirable fish use to migrate, coupled with shallow water depths at the base of the weir to prevent large-bodied fish jumping Photo credit: Cindy Baker.

Gabion basket weir barriers have been trialled as a selective exclusion barrier in two locations in New Zealand, with the aim to provide upstream and downstream passage for desirable species while also preventing undesirable species' access upstream (Figure 6-16 and Figure 6-17). Both did limit some undesirable fish access in the short term, but failed to consistently provide partial exclusion long-term due to silt accumulation causing infilling of the gaps within the baskets intended to provide passage for desirable juveniles, and invasive undesirable macrophytes establishing on the structure (Clucas 2016). These factors resulted in the need for regular cleaning of the structure, and it was found that the physical structure degraded and changed shape over time. In general, gabion basket weirs are not recommended for use as selective barriers because juvenile fish can be damaged from passage through the rock.

Figure 6-16: An unsuccessful gabion basket weir installed to allow migratory native fish access, while preventing trout access in Orokonui Creek, Otago. Photo credit: Sjaan Bowie.

Figure 6-17: An unsuccessful gabion basket weir installed to protect a dwarf galaxias stronghold from trout in an unnamed tributary of the Maruia River, West Coast. Photo credit: Sjaan Bowie.

6.3.7 Hydrological and physical features

There are many natural and manmade features that can be used to aid the design of successful exclusion barriers. Hydrological and physical features of natural waterfalls, overhangs, cascades, wetlands, unfavourable natural conditions (e.g., drying/high water velocities) and physical features of known barriers (e.g., screens, falls) can be combined with biological traits and behaviours to aid successful exclusion barrier design (Table 6-3; Charters (2013)). High water velocities and/or low water depth can create physical features unfavourable to the upriver movements of undesirable fish species. In addition, disturbed or ephemeral reaches of streams have been found to protect multiple non-migratory galaxiids from undesirable salmonids impacts in several locations as these reaches do not favour salmonid persistence (Woodford and McIntosh 2013). However, Hore (2022) found that although low-flow conditions reduced predation on native galaxiids by trout, native fish populations were also reduced by low flows. The findings of Hore (2022) highlight the careful balance that needs to be achieved to ensure native fish protection long term and the importance of ongoing monitoring to ensure barriers with hydrological features are fit for purpose.

All physical and hydrological exclusion barrier features have advantages, limitations and factors that control their effectiveness (Table 6-3) and that need to be considered when they are used as a management tool to protect native values.

Table 6-3: Physical and hydrological features that have been found to barriers to fish passage. Adapted from Charters (2013).

Type of Barrier	Type	Barrier Mechanism	Factors Controlling Effectiveness	Advantages	Limitations
Low water levels/depth	Natural	▪ Prevent swimming, also known to cause stress for fish ▪ Shallow water depth can prevent some species/life stages	▪ Hydraulic and environmental conditions ▪ Permanence of conditions establishing a barrier ▪ Species-specific; what one species can tolerate, another may thrive	▪ Can be an effective selective barrier, especially as these environments can favour some native fish being sustained over undesirable fish	▪ Changes in natural conditions can result in changes in barrier effectiveness
Uninhabitable zone such as swamps, ephemeral stream reaches and dry stream beds	Natural	▪ Species-specific; what one species cannot tolerate, another may be perfectly healthy in ▪ Vegetated channels with lack of surface flow and/or ephemeral flows prevent or limit fish access ▪ Prevents swimming/access to habitat	▪ Hydraulic and environmental conditions ▪ Permeance of conditions ▪ Species-specific; what one species can tolerate, another may thrive	▪ Can be an effective selective barrier, especially as these environments can favour some native fish being sustained over undesirable fish	▪ Changes in natural conditions can result in changes in barrier effectiveness
Dams/ Waterfalls (fall height)	Built & natural	▪ The dam/ waterfall height creates a full or selective exclusion barrier in the waterway (preventing swimming, jumping, or climbing) depending on fish community ▪ Sometimes aprons or other additional add-ons can provide fall height/perch to prevent passage	▪ Height of barrier ▪ Surface of barrier – specific species abilities to negotiate dam structure. Such as if wetted margins or form on dam face allows native fish to climb face ▪ Presence of spillway (weir), fish pass or fish trap-and-transfer facilities ▪ Maintenance	▪ Can be full or selective exclusion barriers dependent on species present ▪ Dams may have been installed for another purpose or waterfalls formed, but exclusion of undesirable species results	▪ Dams can result in significant alteration of stream hydrology, sediment transportation and consequently, in-stream habitats ▪ Dams create large amount of infrastructure and are high cost

Type of Barrier	Type	Barrier Mechanism	Factors Controlling Effectiveness	Advantages	Limitations
Chutes (Velocity)	Built & natural	High water velocity fatigues fish before they can fully negotiate a barrier (i.e., it exceeds their maximum swimming ability (see Section 2.3)). Increased velocities can be achieved in natural cascades in waterways or through placement of a culvert or chute that constricts the water flow. Shallow water depth in or downstream of these barriers can prevent larger fish from swimming as well as inhibit their ability to jump	▪ Flow velocity and depth of water in and/or downstream of chute ▪ Hydraulics during differing flow conditions ▪ Fishes' swimming ability and behaviour	▪ Less hydrological effect than weirs or dams ▪ Can function as selective barriers (i.e., they exclude one species while allow another species passage, particularly for weak-swimming species)	▪ Different fish species have different swimming performances and so their ability to negotiate a velocity barrier varies ▪ Salmonids and trout species are strong swimmers, and therefore velocity barriers may be insufficient to prevent them passing upstream ▪ Changes in natural conditions can result in changes in barrier effectiveness
Falls/weirs	Built	A weir can be used to create a full or selective exclusion barrier by various mechanisms, including a vertical barrier exceeding or preventing undesirable fish access e.g., jumping, or creating a concentrated zone of fast flow over its crest, or downstream or upstream additional structures like aprons	▪ Height of structure crest ▪ Downstream pool that prevent jumping ability ▪ Presence of an upstream pool that alters habitat upstream ▪ Flow velocity and depth ▪ Hydraulics during high flow conditions	▪ Less hydrological effects than dam ▪ Precast components available ▪ Shallow apron that prevents jumping fish	▪ Change in hydraulics under high flow conditions may reduce barrier effectiveness (e.g., raised tailwater depth (pooling at base)) ▪ Instream structures have been known to degrade and deform over time, adversely affecting their performance as a barrier ▪ Instream structures can be a high cost dependent on design required, planning processes and accessibility of site
Screens/grills	Built	Screens physically block biota (including adult and juvenile fish, and fish eggs) over certain sizes from passing through, while allowing water to continue flowing. This	▪ Hydraulics during high flows (e.g., overtopping a gabion basket weir may occur) ▪ Permanence of barrier	▪ Can be an effective full or selective barrier to selectively prevent or allow	▪ Screen, grills and other structures (e.g., gabion baskets) have been known to degrade and deform over time,

Type of Barrier	Type	Barrier Mechanism	Factors Controlling Effectiveness	Advantages	Limitations
		could be gabion basket weirs, that are established to try and let small/ climbing fish but exclude large upstream migrating fish, or fish screens in water intakes, that are established to take water from waterways and prevent entrainment or impingement of fish otherwise they are lost to the fishery, or structures with screens, protruding grills, lips or bars sized and spaced appropriately to prevent access	▪ Water intake design parameters are maintained over time (e.g., approach & sweep velocity, screen material opening gap) ▪ Screens, grills, and bars are maintained on structures to prevent undesirable access	access to particular species ▪ Gabion basket weirs and screens can allow stream flow to continue through barrier, with minimal impact on hydraulics ▪ Barrier to prevent downstream or upstream movement	adversely affecting their performance as a barrier ▪ Need maintenance as can be silted up or clogged with macrophytes/debris ▪ High velocities and conditions at screen and water intake interfaces may trap or harm fish if not designed appropriate for the location and species, and maintained ▪ Instream structures can be a high cost dependent on design required, planning processes and accessibility of site
Overhanging lips	Built & natural	Overhangs can be created by waterfalls, built solid structures (e.g., culverts) or grated or solid lips hanging out from the downstream face of a barrier	▪ Height of the overhang ▪ Width, length (protrusion), spacing (if not solid) and angle of overhang from downstream face ▪ Grate spacing, if not solid overhang	▪ Provides additional barrier against jumping ▪ Exclusion barrier for climbing species	▪ Can block native climbing species ▪ Instream structures have been known to degrade and deform over time, adversely affecting their performance as a barrier ▪ Instream structures can be a high cost dependent on design required, planning processes and accessibility of site

6.3.8 Minimising the upstream effects

Another consideration in exclusion barrier design is minimising any upstream effects on stream hydrology and habitat of the vulnerable species or habitat you are trying to protect. Salant et al. (2012), Bowie et al. (2013), Birnie-Gauvin et al. (2017) and other New Zealand experiences have found that as a result of exclusion barrier installation, riffles and gravel substrate can often be reduced, silt can settle and build up against the upstream weir face, deeper pools can be established upstream of the weir, habitat can change upstream, nutrient levels can increase and algae proliferation can lead to reduced water and habitat quality, and if installed in a shallow gradient area, an extensive area of backwater can establish (e.g., Figure 6-4). These changes may enhance, or more likely reduce, the available habitat of the desirable species upstream, may cause flooding beyond areas previously flooded, and change availability of habitat and the balance of the aquatic community (Salant et al. 2012; Bubb et al. 2021). For many exclusion barriers, minimising the upstream effects are crucial to not reduce the desirable species' habitat.

An indirect effect of a larger backwater that should be considered is the possible increase or establishment of macrophytes in the upstream backwater or surrounding habitats. Macrophyte invasion, such as *Erythranthe guttata*, can reduce fish abundance (Gallardo et al. 2016) and can potentially provide preferential habitat for undesirable salmonids (Lusardi et al. 2018; Marsh et al. 2022).

Where silt and water build-up is expected upstream of exclusion barriers, New Zealand installations have included v-notch or overflow profiles (Figure 6-3), a perched culvert pipe within the weir (Figure 6-5) to allow continual draining, or have included a pipe with stopper or removable boards or slots (Figure 6-4 and Figure 6-18) within barriers that can be used to manually drop water levels and flush sediment build up.

Environment Agency (2009) proposed a formula for estimating backwater length for typical vertical weirs that could be a useful starting point for approximating the potential upstream effects of weir construction:

$$L_{bw} = 0.7\frac{d}{S} \tag{18}$$

Where L_{bw} = backwater length (km), d = water depth (m) and S = stream gradient (m/km).

Figure 6-18: Wooden slots on the Fraser Spring exclusion barrier that can be manually removed to flush sediment and lower water levels (photo showing one removed). Photo credit: Sjaan Bowie.

6.3.9 Design flow

Considerations associated with the design flow are the hydraulic profile over and/or through the barrier under varying flows, anchoring of the structure to prevent overturning, sliding, or scour during high flows, and protection of abutments (Charters 2013). Most barriers use 1:100 year flood flows as the maximum design flow for full exclusion. Defining the expected flood characteristics, stream flow, stage height, flow paths in the vicinity of the barrier and levels of any debris are needed. Consideration may need to be given to whether this needs to be increased dependent on predictions arising from changing climates.

6.4 Good practice design, installation, maintenance, and monitoring requirements for exclusion barriers

Biological and behavioural traits of fish species (Section 6.3.2–6.3.6), as well as hydrological and physical characteristics (Section 6.3.7–6.3.9) all need consideration in the design of successful exclusion barriers. The use of exclusion barriers (<4 m in height) as a management tool to protect key species' locations and habitats is increasing in regularity and will likely be a key protection tool for some of our most threatened fish under changing climate conditions predicted in the future.

Information and lessons learned from exclusion barriers in New Zealand and overseas can now provide some good guidelines for future exclusion barrier use in New Zealand. Most physical exclusion barriers installed in New Zealand have been full exclusion weirs designed to prevent the movement of salmonids, and have successfully resulted in the protection of several key non-migratory galaxiid locations when combined with undesirable species removal operations (Figure 6-2 and Figure 6-5). Some successful selective barriers (Figure 6-11; Figure 6-12; Figure 6-11; Figure 6-14; Figure 6-15) have also been established. However, it should be noted that there are still knowledge gaps, and ongoing monitoring and maintenance is crucial for success and to improve management and future designs (section 8).

For vertical weir exclusion barriers, barrier location, height, profile, flow, and downstream zone have been found to be the key successful design features (Table 6-4). Along with knowledge of the objectives, the undesirable species to be excluded, species that need protection and/or passage, and key site characteristics, a design can be established by working through the key design features (Table 6-4). If the exclusion barrier is planned after undesirable species have already accessed the location, a physical removal programme will also be needed for success.

Table 6-4: Design considerations for (weir) exclusion barriers. Adapted and updated from Charters (2013).

Design Feature	Design Criteria	Design Considerations
Barrier location	▪ Barrier placed in a stable section of streambed, with a moderate slope and small floodplain area ▪ Uninhabitable habitat that will aid preventing passage of undesirable species are used where available (dry stream bed, ephemeral/swamp reaches)	▪ Placed in a hydrologically stable reach ▪ Minimise upstream backwater effects including loss of riffle zones and flooding by placing barrier in section of reasonable gradient or altering barrier slope
Barrier height/ perch (fall height)	▪ Drops ≥1- 1.5 m are effective exclusion barriers to most undesirable fish ▪ Drops <1m should be used in combination with strengthening other barrier features, such as a shallow, high velocity chute, screens, perched overflow, or overhanging lips	▪ Minimising upstream backwater effects by minimising barrier height while still achieving barrier effectiveness ▪ Change in sediment and debris transport within stream ▪ Creating perch/overhang where weir height is not sufficient
Barrier profile	▪ Upstream face angle maximised (90-degree angle preference) ▪ V-notch profiles or perched culverts to maintain a concentrated, high-velocity body of flow under low flow conditions	▪ Minimise upstream backwater effects by using a shallower upstream face profile or locating in a moderate slope area ▪ If a pool upstream is formed, it should be eliminated where possible to increase stability of the structure and removing the habitat for undesirable fish to access ▪ Scour protection downstream and side (wingwalls sloped) of the apron

Design Feature	Design Criteria	Design Considerations
	▪ Existing barriers have used ≥0.5 m overhangs to inhibit jumping and/or climbing ▪ Overflow, drop log or perch pipe features can be added to the barrier face to help manage flow and allow sediment flushing ▪ Grills/screen/bar/lip added to exclude undesirable fish (and pass desirable where appropriate/possible) ▪ Availability of wetted margins and roughened surface to provide desirable climbing species access where appropriate (applicable to selective barriers only)	▪ Smooth surfaces to prevent passage; roughened surface when providing desirable passage ▪ Grates/overhangs have been used to prevent undesirable species when height cannot achieve exclusion ▪ Grills/screen allow certain size swimmers to pass up through barrier
Design flow (hydraulic profile)	▪ Existing barriers (in the US) have used 1:100-year flood flows as the maximum design flow for full exclusion ▪ Ensure provides flows that prevent undesirable species (and provides passage for desirable species where appropriate/possible) over full range of flows	▪ Hydraulic profile over weir crest under varying flows ▪ Anchoring of weir structure to prevent overturning, sliding and scour ▪ Protection of abutments ▪ Minimise favourable conditions for undesirable macrophytes establishment upstream ▪ Flow velocities and depths in and downstream of structure ▪ Consider any expected future change in hydrological regime
Downstream zone	▪ Downstream apron (>2 m length) to eliminate pooling and create a high velocity and shallow water zone that inhibits jumping and swimming	▪ Scour protection on sides and downstream of apron to ensure integrity of structure maintained long term and eliminate any opportunity for bypass around structure ▪ Rocks may need to be removed from the downstream area to reduce areas of slow water and ponding (e.g., increase water flow away from downstream side of barrier, eliminate back eddies off the rocks)

Successful exclusion barriers have typically included some of these key features:

- Drops >1–1.5 m. However, if this fall height is not possible, increased focus must be placed on incorporating other features such as overhangs, screens, or non-physical barriers (e.g., shallow, high velocity water) to compensate for lower fall heights.
- Barrier face slope of 90 degrees has been most effective, and any lesser angle must be balanced against drop and other features that will still prevent undesirable species access.

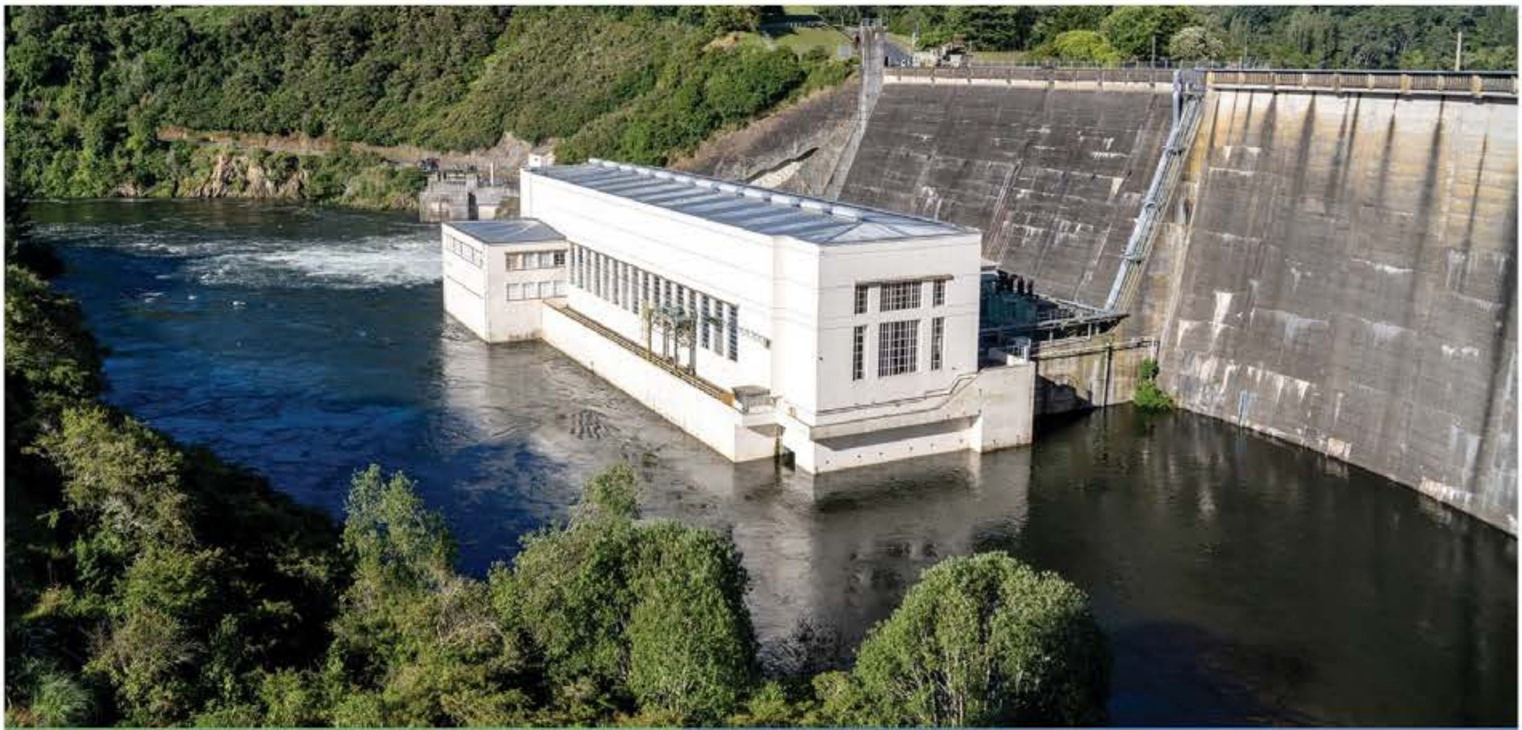
- Downstream apron >2 m in length that creates an area of fast water velocity and low water depth to inhibit undesirable species jumping.
- Upstream backwater effects are minimised by setting the barrier within a stream reach with reasonable slope or, where this is not possible, including additions that provide further perching (e.g., perched culvert within barrier face) and or flushing. Substrate or other structures could also be added to establish and maintain shallow habitat (e.g., add large rocks or a concrete pad).
- Scour protection downstream and to the sides of the apron to cater for any hydraulic jump that may form, protection in high flows, and generally ensure the structure's integrity will be maintained over time.
- The barrier should be located where the channel is stable with a moderate slope to provide drop/fall.
- Waterways in highly erodible soils, steep stream beds and/or made up of very mobile substrates should be avoided where possible due to high erodibility and likelihood of barrier integrity being compromised over time.

Additional criteria that are worthwhile considering include:

- If silt and water build-up upstream is of concern then a v-notch profile, drop log structure, a perched culvert, or a culvert pipe with stopper within the weir could be considered to provide for flushing and/or maintain a concentrated high water velocity under low flow conditions if required.
- Overhangs could be added to physical structures to inhibit jumpers and or climbers (> 0.5 m), especially where fall height cannot be maximised to prevent jumping undesirable species.

The importance of these different design criteria varies depending on the species being excluded, species and habitat being protected and the general environment. Thus, it is important to understand the objective of the barrier before finalising any design and to obtain input from relevant experts on appropriate designs.

Regular maintenance and ongoing monitoring are crucial for all exclusion barriers to ensure the key barrier features are maintained, to confirm undesirable species are prevented and desirable species are secure and protected. Some exclusion barriers, especially those that only prevent access of some life stages, will require on-going effort and resourcing, while other barriers are likely to need maintenance and monitoring checks predominately after significant natural events that could threaten the barrier integrity.



Fish passage at dams

7 Fish passage at dams

7.1 Background

Thousands of dams exist in New Zealand for various uses, including irrigation, flood control, hydro-electricity generation, as well as domestic and industrial water supply (Figure 7-1 and Figure 7-2). While most are small, low-head water supply farm dams with heights of 2–5 m, more than 400 dams have storage capacities exceeding 30,000 m^3 of water, with some reaching heights of over 110 m (e.g., Benmore hydro-electric dam on the Waitaki River; Pickrill and Irwin (1986)). The construction of dams has played a vital role in New Zealand's socio-economic development, yet it has also resulted in significant impacts on New Zealand's diadromous aquatic communities (Baxter 1977; Jellyman and Harding 2012).

In this section, we adopt the definition of a dam according to Section 7 of the Building Act 2004, which classifies a dam as an artificial barrier that spans a stream or river:

- at a height greater than 4 m, holding over 20,000 m^3 of water

or

- at a height lower than 4 m, at or above 30,000 m^3 in volume.

Furthermore, we broadly define the term 'dam' to include any type of barrier that crosses a river or stream channel with the function of impounding or diverting water. These structures obstruct the natural free flow of water, the natural passage of fish, sediment, and other essential nutrients in river system. Dams can be constructed using various materials, including but not limited to, concrete, stone, brick, and earth (The Nature Conservancy 2022).

Instream structures that dam or divert a natural waterway (e.g. large dams) are subject to the requirements of Regulations 43–50 of the FFR83, in addition to relevant NPS-FM and regional plan rules. It is an offence under the FFR83 to propose to build such structures without dispensation from DOC or an approved fish facility. For any such structure that was built post-1983 that has neither dispensation nor an approved fish facility:

- If you were the builder/authoriser, DOC can issue you with a dispensation approving the lack of fish facility, or a requirement to build an approved fish facility.
- If you are not the builder/authoriser (i.e. you are a subsequent landowner) you can get a letter of assurance, or a letter stating that DOC would like you to build a fish facility.

7.1.1 Scope of this section

This section discusses some of the primary means of providing upstream and downstream fish passage at dams >4 m high in New Zealand. These larger structures present similar challenges to small structures although on a much larger scale. There are few effectively functioning technical fishways and/or bypasses in New Zealand, and advances in the design of fishways for structures >4 m in New Zealand are limited. Consequently, much of the design criteria and supporting research is from work in the Americas, UK, Europe, and Australia.

Because of the site-specific nature of fish passage design at large infrastructure projects, we are not able to provide specific design guidelines for fish passage at large structures in New Zealand.

However, **for all new dams, regardless of height or size, provision of effective and efficient upstream and downstream fish passage for all species must be an integral part of the structural and operational design**. **This will require that fish passage is incorporated into the scoping and design process (including costing) from the outset**, rather than being dealt with retrospectively after other aspects of the structural design have been agreed (Williams et al. 2012). It is essential that this includes financing of and collaboration between recognised experts in fish biology and ecology, ecohydraulics, hydraulics and engineering from the outset and throughout the project. The absence of this expertise from the outset of the project will inevitably lead to costly failures in design and implementation. As a general rule-of-thumb, international experience indicates that fish passage design and construction will typically account for at least 10% of total project costs for a new structure. Furthermore, maintenance and operation of fishways (and potentially ongoing adaptive management and refinement) should be included as part of the ongoing operating costs for the structure.

For clarity, we discuss upstream and downstream fish passage solutions in separate sections. It is important to understand, however, that some systems, like trap-and-transfer and nature-like fishways, are inherently bidirectional. Despite this dual functionality, these systems will be addressed in both sections, aligned with their primary intended use.

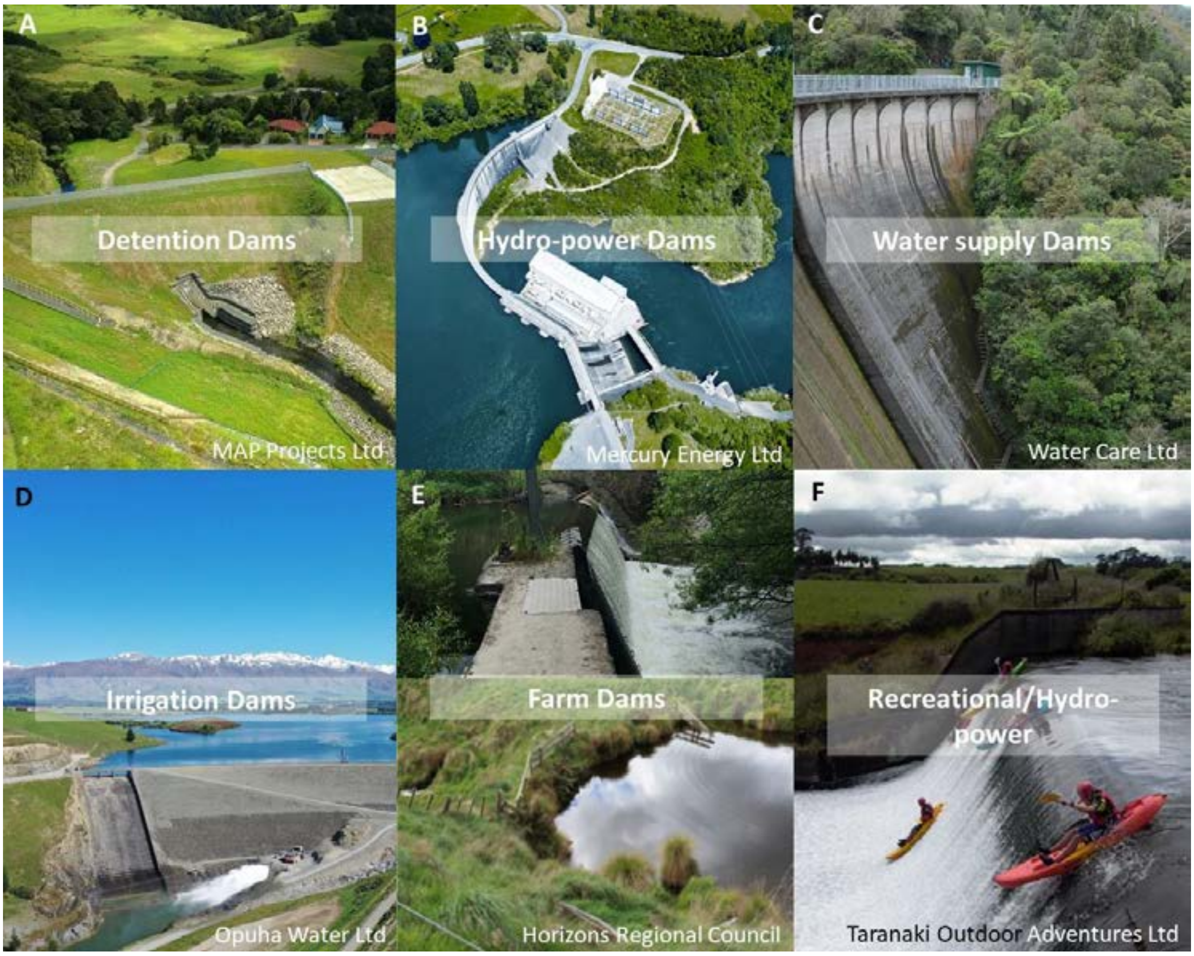


Figure 7-1: Examples of different dam types installed across New Zealand. A = Hopua Dam, Northland; B = Karāpiro Dam, Waikato; C = Upper Nihotupu Dam, Auckland; D = Opuha River Dam, Canterbury; E = Manawatū -Whanganui; Waingongoro River Dam, Taranaki.

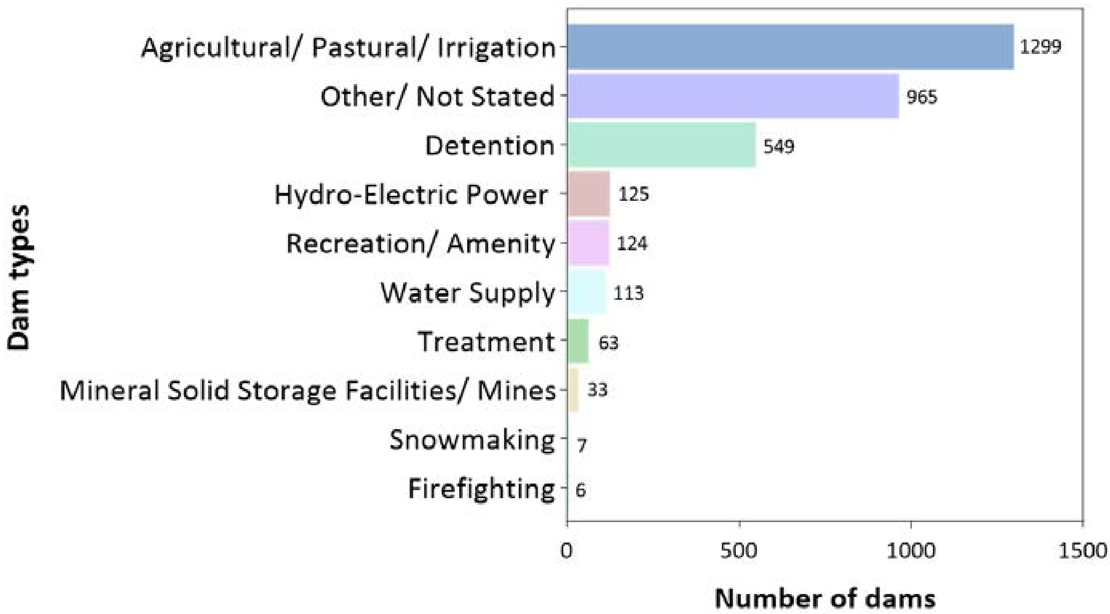


Figure 7-2: Number of known dams in New Zealand by type of use. Adapted from Proposed regulatory framework for dam safety (mbie.govt.nz) (2019).

7.1.2 Dam impacts on fish communities

Dams alter streams and rivers by reducing connectivity, creating artificial lakes, and disrupting the natural processes in sediment transport and seasonal variations in water temperature and water flow. These changes can significantly affect native and sports fish communities. However, because each dam presents a unique case, the extent of each impact varies, depending on several factors including the design and operation of the dam, the characteristics of the local environment (i.e., the stream or river and its fish communities), and the effectiveness of any implemented mitigation strategies (Nielsen and Szabo-Meszaros 2022).

One of the most significant consequences of dams is the obstruction, or delay, of critical upstream and downstream movements and migration pathways of native diadromous fish species, which rely on free-flowing waterbodies to access crucial feeding and spawning habitats to complete their life cycle. These consequences are often further compounded by a range of hydraulic, morphological, and thermal changes to their habitats (Figure 7-3). Such impacts often lead to ecological shifts, reduced biodiversity, and altered aquatic community structures that can ultimately result in the extirpation of species (Jellyman and Harding 2012; Loures and Pompeu 2018; Nielsen and Szabo-Meszaros 2022).

Upstream migrating fish species may be directly impacted by barriers due to the high energetic costs of having to overcome the physical barriers and swimming against strong currents created by the dam. Moreover, obstructions can cause injury and even mortality as fish attempt to navigate them. Delays, disorientation, and injuries can leave fish vulnerable to predation (Nyqvist et al. 2017). Fish needing to migrate downstream face similar consequences; dams generally completely block passage downstream, except when fish move through turbines or spillways at hydro-electric facilities (if no passage over the crests is provided). Here, high risk of injury and mortality occurs through impingement on trash-racks and screens, and during movement through turbines (e.g., blade strike, barotrauma, cavitation, and shear) or from passing over spillways (Čada 2001; Kemp 2015; Algera et al. 2020).

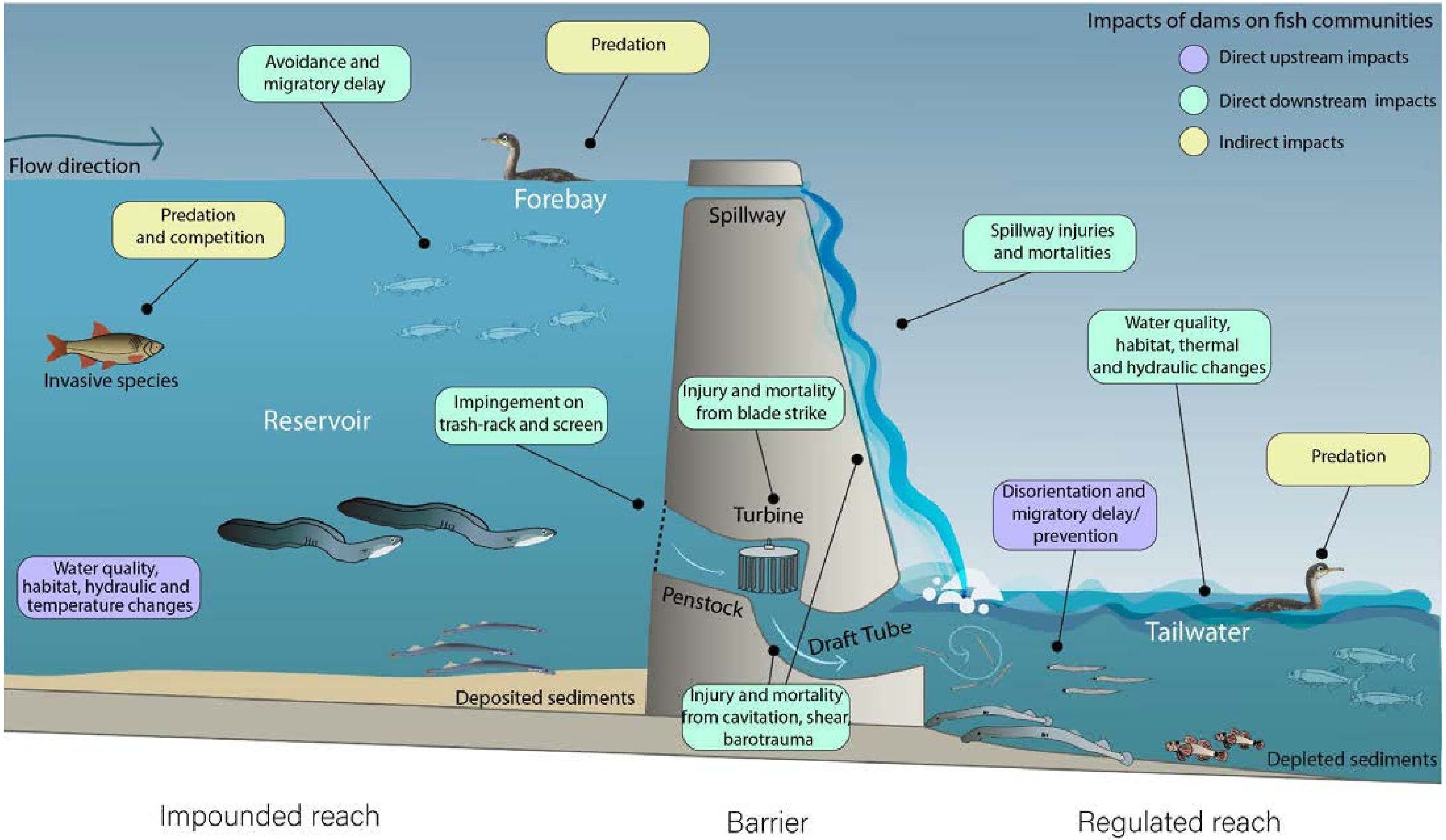


Figure 7-3: Direct and indirect impacts of dams on upstream and downstream fish communities. Credit: Michele Melchior

Lastly, impounding a stream or river and transforming it from lotic (flowing water) to lentic (still water) conditions, often leads to changes in fish communities, and increases the proliferation of exotic fish species. This, in turn, increases pressure on native fish from competition and predation and further alters aquatic community structures, exacerbating the negative impacts of dams on native fish populations (Jellyman and Harding 2012; Algera et al. 2020; Nielsen and Szabo-Meszaros 2022).

7.2 Dam removal

While not all dams are suitable for removal, when the ecological objective is to re-establish connectivity for whole fish communities, dam removal will almost always be the most optimal and sustainable approach. Once dams are removed, rivers can recover substantially from the impacts of damming (Duda et al. 2021). However, it is crucial to recognise that the resulting ecosystem structure and function may be very different to what existed prior to dam emplacement. Consequently, it is essential to understand the potential range of ecological responses to dam removal, including recovery trajectories and future conditions that are likely to occur. Aligning objectives with the anticipated recovery trajectories allows managers and practitioners to set more realistic goals and develop appropriate management strategies (Bellmore et al. 2019).

As dams age and reach their average design life of approximately 60–100 years, many no longer serve their intended purpose and can become less economically and operationally viable (Ho et al. 2017; Belletti et al. 2020). The costs associated with regular maintenance, upgrading machinery to meet regulatory requirements, and potential liability risks may outweigh the economic benefits (Habel et al. 2020). Consequently, the trend towards dam removal is growing globally, becoming an increasingly important management strategy in the United States and Europe. This shift is evident in the increasing number of dam removals, with 2022 seeing 325 barriers removed in 16 European countries (Magilligan et al. 2016; Sala and van Treeck 2021; Mouchlianitis 2022).

7.2.1 Dam removal process

The process to remove large barriers such as dams is complex and requires broad stakeholder consultation to ensure representation from environmental, social, economic, and cultural perspectives. In addition, detailed risk assessments are essential to understand potential site-specific impacts, and for sediment control and riparian rehabilitation (O'Connor et al. 2017b; Bellmore et al. 2019). For instance, dams may hold back significant amounts of impounded sediment and contaminants and removing them without proper planning can result in downstream sedimentation, and potential damage to habitats. Furthermore, it may take some time to establish a channel within the old reservoir and it may be a long-term process before a stable pattern and profile becomes established. Barriers are also important for maintaining isolated populations (i.e., protection from predators, non-native species and genetic isolation (Doyle and Harbor 2003; Stanley and Doyle 2003), which needs to be evaluated and planned for before dam removal is considered (see Section 6 for further details).

Dam removal may involve the complete or phased/partial dismantling of structures to reinstate the natural open channel allowing all species and sizes of fish and other aquatic species to move and migrate (O'Connor et al. 2017b). During complete and partial dam removal, natural channel design techniques such as installing rock riffles (Section 5.5) or a series of large boulders can stabilise accumulated sediments, provide habitat for fish, reduce water velocities, improve downstream water quality (e.g., dissolved oxygen) and advance recovery towards a more stable and ecologically diverse stream or river. Under certain circumstances, partial removal may be more advantageous, for example, if full removal would release a significant amount of sediment (Katopodis and Aadland 2006). For detailed guidelines and best practices in dam removal the following resource is recommended: [A practitioner's toolkit for dam removal - Dam Removal Europe].

7.2.2 Ecological changes following dam removal

Immediate impacts on fish communities are commonly observed following the removal of dams (Duda et al. 2021). Several studies conducted in the United States (<3 years after dam removal), such as those in the Pine River (Michigan; Burroughs et al. (2010)), Baraboo River (Wisconsin; Catalano et al. (2007)), Rappahannock River (Virginia; Hitt et al. (2012)), and Penobscot River (Maine; Case Study 7; Watson et al. (2018a)), have provided key insights into short-term ecological changes (Watson et al. 2018a). The studies have documented the recolonisation of diadromous fishes in newly accessible habitats (Hitt et al. 2012; Hogg et al. 2015), increased diversity upstream of former dams (Burroughs et al. 2010), and the integration of marine-derived nutrients and energy into food webs (Tonra et al. 2015). However, the complete effects of river modification on fish communities often become more evident over longer time scales, exceeding 10 to 20 years (Kruk et al. 2016; Whittum et al. 2023).

Case Study 7: Penobscot River Short- and Long-Term Success Post-Dam Removal

The Penobscot River Restoration Project is an example of short-term and long-term success following dam removal. In Maine, diadromous fish populations had suffered significant declines due to loss of accessible habitat (Lake et al. 2012), in combination with overfishing, pollution and climate change. Beginning in 2012, the removal of the two lowermost dams (the Great Works Dam [6 m height], followed by the Veazie Dam [6.1 m height] see Figure 7-4) opened 15 km of main-stem river access, leading to significant changes in fish assemblages.

A comprehensive survey was conducted over a period of 3 years prior to the rehabilitation, 3 years after the rehabilitation, and 8 years after the rehabilitation. The most pronounced shifts in assemblage structure were observed immediately after the dam removal in previously impounded sections. These areas experienced an increase in the presence of riverine and migratory species. Over the long term, changes were also documented in tributaries and river segments influenced by tides, where there was a notable rise in the abundance of adult and young-of-the-year river herring (*Alosa* sp.; (Whittum et al. 2023). This case study highlights the long-lasting positive impacts of dam removal on fish communities, with observable changes occurring immediately after removal and over an extended timeframe.

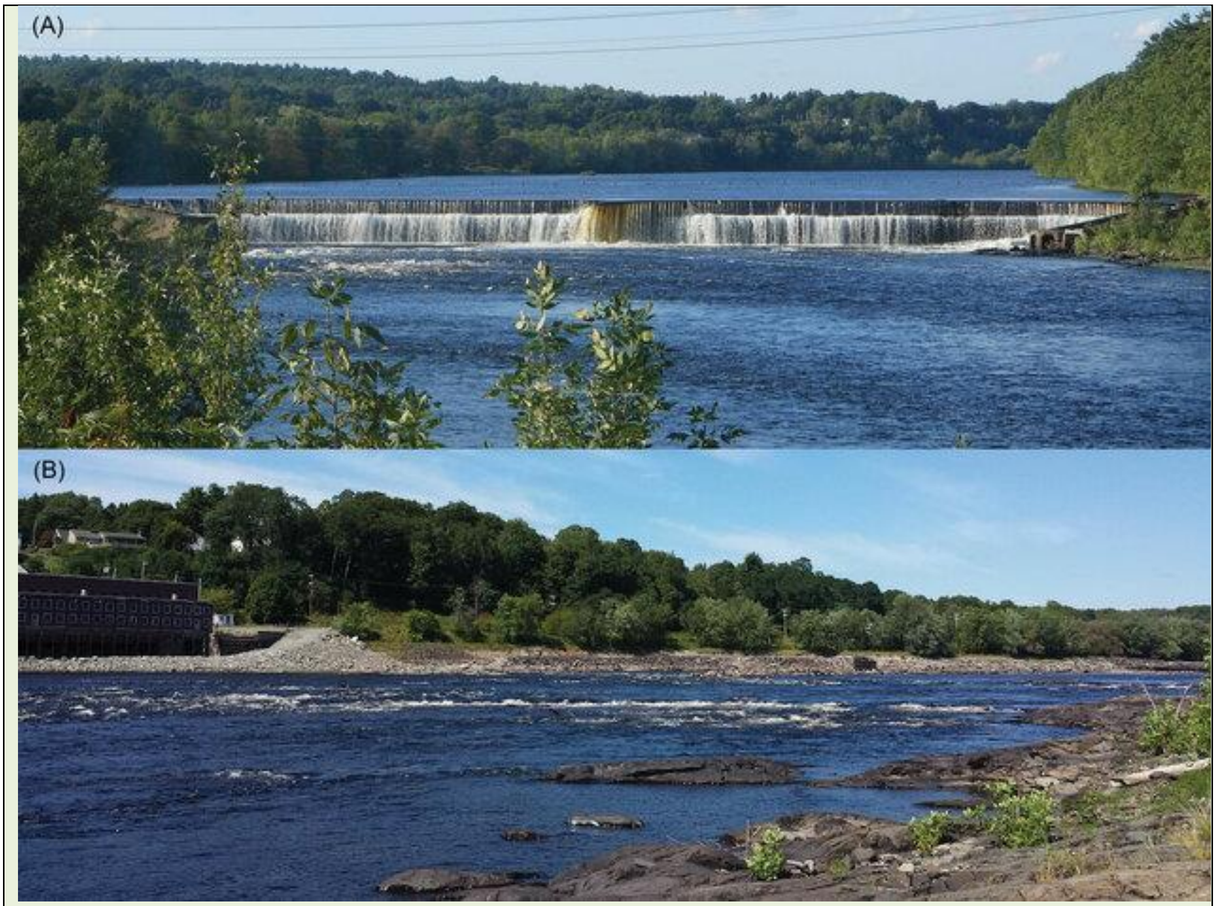


Figure 7-4: The Veazie Dam and Penobscot River before (2012) and after (2013) removal. Reproduced from Sneddon et al. (2022).

7.3 Solutions for upstream passage

If a dam cannot be removed, modifications can be made to allow fish to pass upstream. The need to provide upstream passage facilities for ensuring long-term sustainability of migratory freshwater fish populations is now well recognised. In determining the appropriate solution for upstream passage, clear objectives and defined performance measures should first be established (Section 3).

Upstream passage facilities can be divided into two groups:

'Non-volitional measures', where fish are transported past high-head structures using mechanised methods. The principal non-volitional measures are locks and lifts, as well as trap-and-transfer.

or

'Volitional measures', where fish choose whether to enter and then pass through the structure. These solutions include technical fishways such as pool-type, baffle type, nature-like fishways, emerging technology fishways, and climbing species fishways. These fish passage structures are generally used at low-head dams (up to 6 m) and complement the migration behaviour and swimming capability of fish to facilitate passage.

In this section, key examples of non-volitional and volitional upstream passage solutions available internationally and in New Zealand are discussed.

7.3.1 Key criteria for designing successful fishways at high head structures

Regardless of the type of fish pass or facility installed at a dam, there are five interlinked, key criteria for ensuring effective passage of fish:

1. **Entrance location**. It is critical that the entrance is located at the upstream limit of migration for the target fish species (Figure 7-5). Failure to locate the entrance correctly will have a significant negative effect on passage efficiency. Section 5.5.3 details how to determine the upstream limit of migration for an instream structure.
2. **Tailwater range**. It is essential to determine the tail water range (i.e., the water levels downstream of the structure and fishway) across the range of flows that the fishway will be designed to accommodate (operational flow range). The expected tail water depths at the operational flow range will determine the appropriate depth of the fishway entrance. The operational flow range will also influence the upstream limit of migration and siting of the entrance location. In some instances, a high and low flow entrance location for the fishway may be necessary.
3. **Fishway dimensions**. Correct sizing of the fishway pools/cells for all target fish species is crucial for passage success. Turbulence, water depths and water velocities will be contingent upon the gradient and size of the fishway cells, and all three factors can negatively influence different fish species' behaviour and migratory motivation. The expected biomass of fish passing through the fishway will also determine the minimum size of the structure. However, the size of the fishway influences the volume of water required and this can be a trade off with the target biomass being catered for. See Table 7-2 and Table 7-3 for guidelines on appropriate fishway dimensions.
4. **Headwater range**. The headwater range (i.e., the water levels at the most upstream point of the structure) will influence the water depth in the fishway and will determine the operational range of flows the fishway will need to accommodate. It is important to accurately determine the headwater and tailwater range before designing the fishway as this will enable the entrance location to be sited to maximise attraction and passage efficacy for the target species.
5. **Exit location**. It is vital to locate the exit of the fishway where fish can safely access the river and continue moving upstream (e.g., Figure 7-5). If the exit is located too close to a spillway or within a high velocity area of the downstream flow, fish could be washed back downstream. Where an intake is present, the exit of the fishway should be located well upstream to ensure fish do not entrain into the intake or impinge upon the screens.

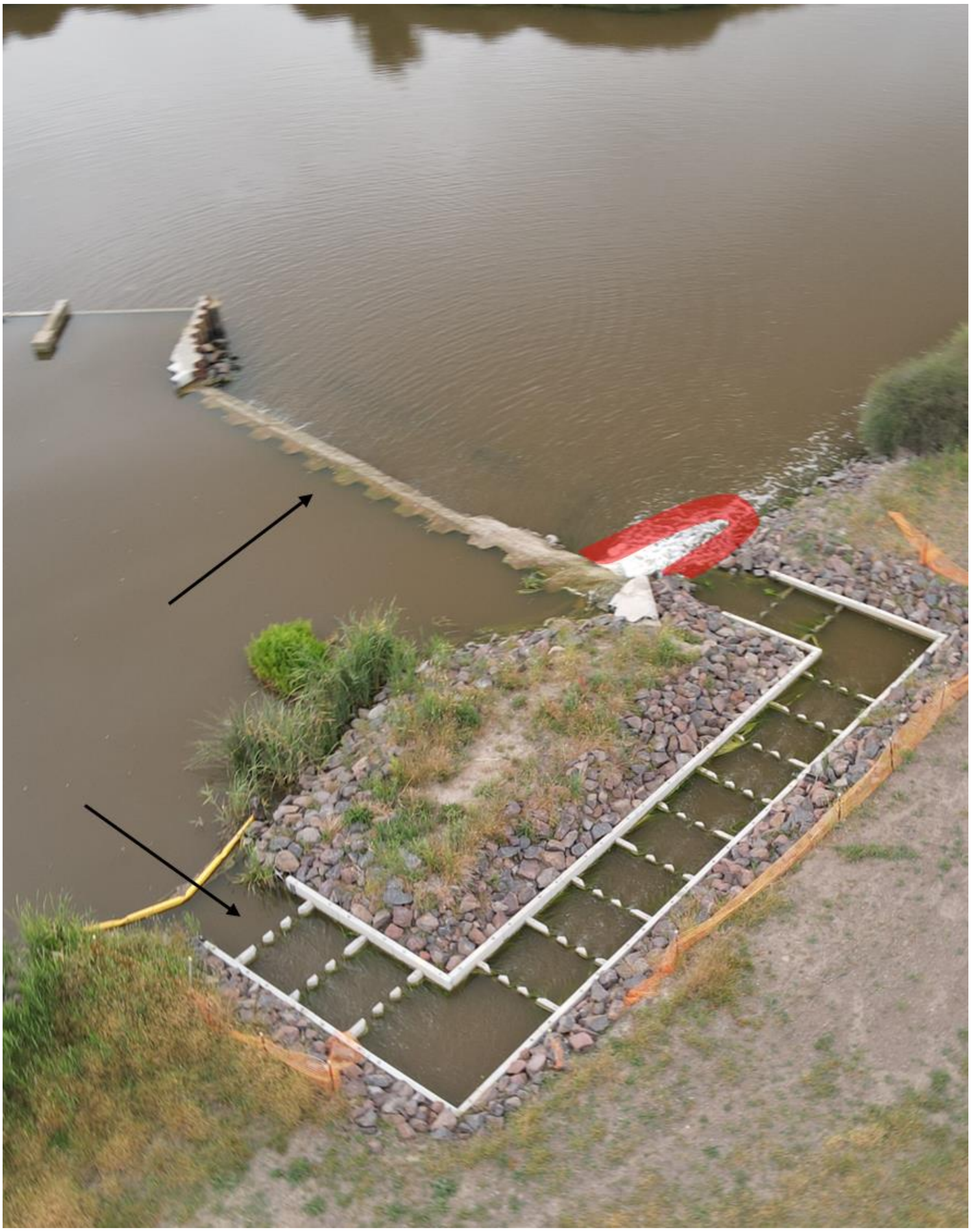


Figure 7-5: Cone fishway. The entrance is located at the upstream limit of migration (red shaded area) and the exit is located well away from the weir crest. Black arrows indicate flow direction. Photo credit: Tim Marsden, Australasian Fish Passage Services.

7.3.2 Upstream passage solutions overview

There are a wide range of approaches available for trying to re-establish or improve the upstream passage of fish past dams. The most appropriate solution in any given context will vary depending on the nature of the structure, local site constraints, the fish communities for which passage is to be provided for, plus social, cultural, as well as economic values and constraints.

Table 7-1 provides an overview of some of the key upstream fish passage facilities available globally and summarises some of their key advantages and disadvantages. The remainder of this section provides more detail on the applicability and deployment of these different solutions in a New Zealand context.

Table 7-1: Overview of key upstream fish passage facilities used internationally and in New Zealand, and their general application described in this section. Adapted from Nielsen and Szabo-Meszaros (2022).

Solutions for upstream passage	General applications	Advantages	Disadvantages
Trap-and-Transfer - Fish ramps, Eel ladders and Lamprey Passage Structures (LPS (Section 7.3.3)	▪ Not limited by the head, applicable for cascade systems (e.g., a series of interconnected dams along a river)	▪ Suitable for juvenile eels, lampreys, and some climbing fish species ▪ Potential to transport fish across any size or multiple dams, with relatively low capital costs ▪ Relatively inexpensive and with low maintenance ▪ Relatively small footprint ▪ Can manage variable tailwater heights and can operate independently of headwater levels ▪ Can operate with minimal flow (e.g., 1 l/s) compared to technical fishways or bypass channels	▪ Requires transport infrastructure between the dam's base and upstream impoundment(s) ▪ Labour-intensive, and potentially high operating costs ▪ Often poor understanding of efficiency ▪ Can require different substrates to be effective for different target species ▪ Species (juvenile eel or climbing lampreys) specific particularly at large dam heights ▪ Often poor understanding of efficiency for different species
Fish lifts and locks (Section 7.3.4)	▪ Scalable to suit the head	▪ Suited to a wide range of fish species and sizes in New Zealand ▪ Ability to lift fish across any dam size with limited space requirements ▪ New technologies are promising based on the first tests	▪ Complicated and expensive technology ▪ Requires power to operate ▪ Will require continuous supervision or daily inspection and adequate maintenance
Technical fishways (Section 7.3.5)			
▪ **Pool-type** (Pool and weir)	▪ Low- to medium-head applications (<3–4 m)	▪ Simple design ▪ Widely used internationally ▪ Wide functioning range but limited effective range	▪ No data to support applicability for New Zealand species ▪ Limited headwater range ▪ Expensive due to concrete channel

Solutions for upstream passage	General applications	Advantages	Disadvantages
▪ Vertical slot fishway	▪ Low to medium- head applications (<6 m)	▪ Widely used and well-known ▪ Relatively flexible headwater range ▪ Design can be adjusted to suit differing channel and fish sizes ▪ Low maintenance ▪ Can achieve relatively high passage efficiency ▪ Operate over a wide range of flow conditions	▪ Little research on New Zealand species but has been shown to successfully pass īnanga, lampreys (*Geotria australis*; Petromyzontidae sp.) and shortfin eels (*Anguilla australis*) ▪ Moderate footprint ▪ Relatively expensive
▪ Trapezoidal	▪ Low-head applications (<3 m)	▪ Moderately inexpensive ▪ Prefabricated design ▪ High and low discharge zones ▪ Suitable for a range of fish sizes	▪ No data to support applicability for New Zealand species ▪ Potentially poor performance for small-bodied fishes
▪ Cone fishway	▪ Low-head applications (<3–5 m)	▪ Prefabricated baffle, relatively easy to construct ▪ Flexible headwater range depending upon design dimensions ▪ Low maintenance, lower average velocity compared to vertical slot fishways ▪ Proven effective for very small bodied (<20 mm) fish in Australia	▪ Limited data available on the performance globally and in New Zealand
▪ **Baffle-type** (Denil fishway)	▪ Can pass benthic species, better suited to fish >40 mm long (slope ca. 1:12) ▪ Low-head applications (<5–10 m)	▪ Relatively cost-effective ▪ Can be used for steep applications with limited space (up to 1:6 slope) ▪ It is more slope tolerant than vertical slot fishways ▪ Can be suitable for benthic species	▪ No data to support applicability for New Zealand species ▪ Design favours larger, strongly swimming fish ▪ Limited headwater range ▪ Evidence of poor passage for some surface migrating and smaller fish

Solutions for upstream passage	General applications	Advantages	Disadvantages
Nature-like fishways (Section 7.3.5)			
▪ Bypass channel	▪ Potentially scalable to suit the head	▪ Resembles natural stream ▪ Capable of passing a broad range of species ▪ Fits into natural surroundings ▪ Provides riverine habitat ▪ It can be constructed from local materials ▪ Not limited by dam height, only footprint size ▪ Suitable for all fish sizes	▪ Expensive ▪ Variable headwater can limit operation unless combined with technical passage types at the hydraulic inlet ▪ Can require significant flows ▪ Low slope usually limits the use of such channels to low-head dams or diversion weirs (<1:30 slope)
▪ Rock ramp	▪ Low-head applications (<3–5 m)	▪ Relatively inexpensive and broad water level operating range ▪ Partial to full-width entry ▪ Provides a riverine habitat ▪ Full width manages wider headwater variation ▪ Suitable for small fish due to channel roughness	▪ Only operates above the whole supply level when there is no spill across the dam ▪ Limited headwater range ▪ Less effective for small fish at high flows
Novel solutions* (Section 7.3.6)			
▪ Whooshh and Tube fishways	▪ Scalable to suit the head height	▪ Ability to lift fish across any dam size with limited space requirements	▪ Unproven technology ▪ Limited data available on the performance globally and in New Zealand

7.3.3 Trap-and-transfer

Trap-and-transfer (also referred to as trap-and-transport or catch-and-carry) is currently the primary means of providing upstream fish passage at high-head structures (primarily hydro-electric schemes) in New Zealand (Figure 7-6). This solution relies on the effective capture of as many of the target species and/or life stages as possible and their manual transportation above barriers.

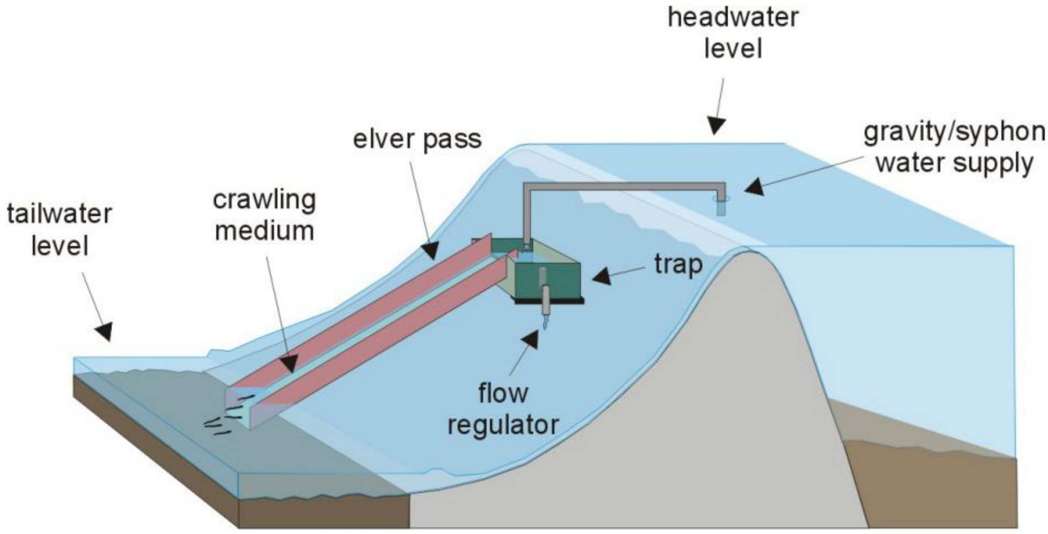


Figure 7-6: Schematic diagram of a trap-and-transfer targeting eels. Reproduced from Solomon and Beach 2004.

Trap-and-transfer has been widely adopted by asset owners in New Zealand because it was deemed practical and cost-effective, especially in the absence of effective permanent upstream passage solutions and because most dams have been constructed without consideration to fish passage. Despite the range of species for which passage should be provided, trap-and-transfer activities in New Zealand are currently primarily focused on juvenile longfin and shortfin eel, and pouched lamprey (*Geotria australis*) (Figure 7-7; Case Study 8).

Customised traps of various configurations and specifications (typically box traps with a ramp and climbing substrate) are manually deployed at the most downstream barrier. Trap designs vary with the specific needs and logistical constraints associated with each site (Crow et al. 2020). The traps are typically emptied daily, with the target fish species removed from the holding box, counted, and transported upstream of the barrier where they are manually stocked into various habitats and locations. A series of modifications to elver traps have been undertaken with the intention of improving fish passage efficiency across various hydro-electric sites (Crow et al. 2020), but there is little quantitative data to evaluate trap efficiency or support these modifications, and there is no national guidelines on elver trap design.

Protocols are generally in place to minimise stress on fish during capture and transport and there are standardised reporting requirements and data recording sheets for elver trap-and-transfer operations (Crow et al. 2020). Trap-and-transfer activities typically quantify the total number of a fish species captured and transported upstream of hydro-infrastructure. Species composition, abundance and recruitment are all monitored and reported, and these data contribute to understanding eel recruitment patterns in New Zealand (Crow et al. 2020).

Case Study 8: Iwi/hapū-driven upstream tuna passage activities at hydroelectrical power stations

The social, cultural, spiritual, economic, and environmental impacts of hydroelectric power stations (HEPS) are enduring for Māori (Waitangi Tribunal 2012; Whetu and Whetu 2019) (Stewart-Harawira 2020). For many, these rivers represent living ancestors, to which they had long held customary rights (Te Aho 2010), and the damming of ancestral rivers for HEPS development was generally undertaken without consultation with iwi and hapū. Iwi and hapū have long expressed the impacts that HEP has had on their health and wellbeing, through tangible and significant losses of lands, water, wāhi tapu, wāhi tupuna, and taonga, including tuna (Waitangi Tribunal 1993; Young et al. 2004; Cunningham et al. 2016).

Many iwi and hapū consider that on-going consenting and planning regimes give preference to HEPS at the expense of ecological and cultural values (Waitangi Tribunal 1993; Durette et al. 2009; Waitangi Tribunal 2012). For example, the Ngāti Manawa Claims Settlement Act 2012 states that "the Ngāti Manawa tuna fishery has been depleted through policies and actions of the Crown including the construction of dams and the favouring of trout fishing over the customary fishery" and "the degradation and development of the Rangitaiki and Wheao Rivers, their tributaries and wetlands have resulted in the decline of its once rich tuna and other fisheries, which for generations sustained the people's way of life and their ability to meet their obligations of manaakitanga, and that the decline has been a further source of distress to Ngāti Manawa". The Ngāti Manawa Claims Settlement Act 2012 includes the following clause around tuna habitat "All persons exercising functions and powers under the Resource Management Act 1991 that affect the Rangitāiki River must have particular regard to the habitat of tuna (*Anguilla dieffenbachii* and *Anguilla australis*) in that river", which includes those areas upon which tuna depend in order to meet their requirements for spawning, rearing, food supply, and migration (New Zealand Government 2012).

There is a long history of failed attempts at providing effective upstream passage for elvers at HEPS in Aotearoa New Zealand that can be attributed to poor design and/or maintenance. Iwi/hapū have played an important role in advocating for and/or informing the design of improved upstream fish passage solutions for tuna at HEPS. Increasingly, iwi/hapū/Māori entities are being engaged and/or subcontracted by HEPS operators to undertake upstream tuna passage mitigation activities.

The provision of upstream elver trap-and-transfer is now well established as a mitigation activity at many large HEPS, although passage efficiency remains poorly quantified at many sites (Martin and Bowman 2016; Crow et al. 2023). There is an increasing number of studies being conducted to help assess the effectiveness of elver trap-and-transfer activities, several of which are focused on the outcomes iwi/hapū are seeking (Smith et al. 2009; Williams et al. 2018; Boubée et al. 2022). However, in the absence of effective and safe downstream migration pathways for adult tuna at most HEPS, the long-term impact of these activities on the sustainability of tuna populations remains unclear (Boubée et al. 2001; Jellyman and Harding 2012).

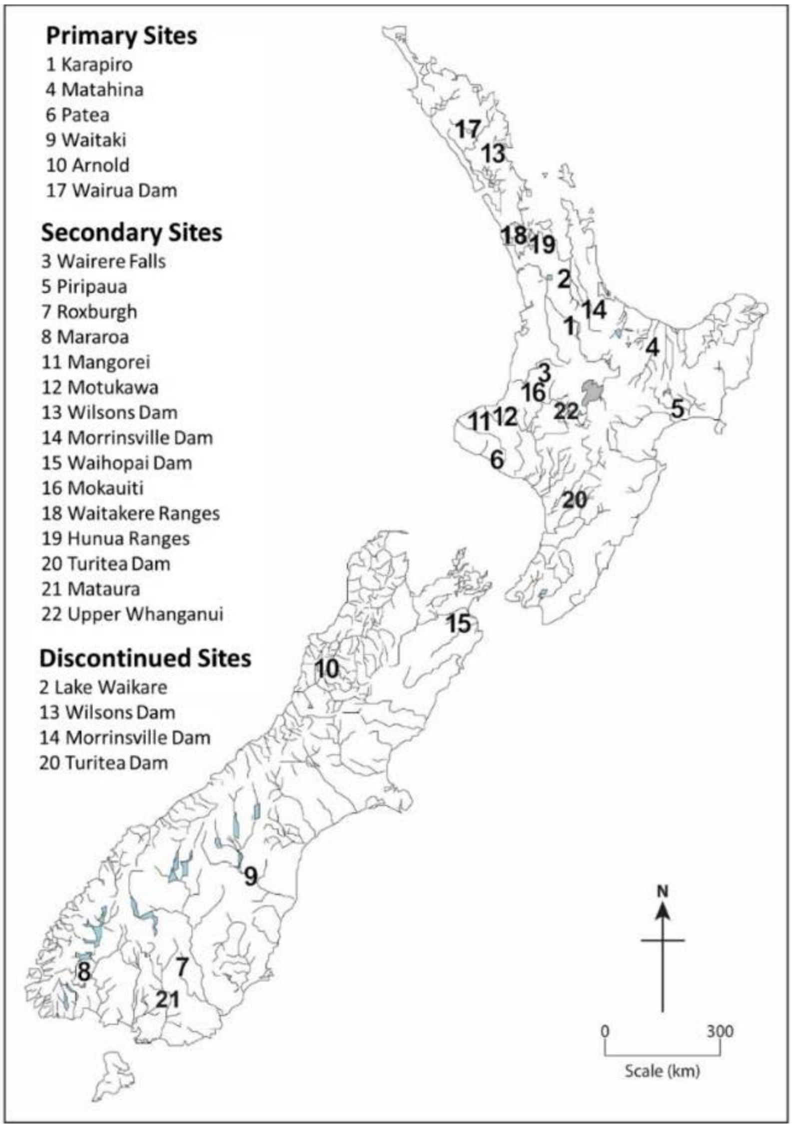


Figure 7-7: Map showing location of elver trap-and-transfer activities at hydro-infrastructure throughout New Zealand. Reproduced from Crow et al. (2020).

Evidence quantifying the performance of trap-and-transfer as a fish passage solution is generally lacking in New Zealand. Undoubtedly these activities serve to mitigate the barrier impacts of hydro-infrastructure on upstream eel and lamprey passage, but trap efficiency has generally not been evaluated and there are no established post-stocking monitoring programmes. For example, implementation of a trap-and-transport programme greatly increased the number of shortfin and longfin elvers upstream of two dams on the Waikato River (Jellyman and Arai 2016), but studies demonstrating success are rare and/or the data are not readily available. Efforts have also been made to assess the effectiveness of pouched lamprey trap-and-transfers upstream of Roxburgh Dam (Ryder 2018) and Patea Dam (Goldsmith 2021) but despite intermittent transfers of adult lamprey upstream of both dams, no larval lamprey have been identified above either dam. Adult lamprey are known to use pheromones released by larvae as a key migration cue (Johnson et al. 2015) and effective trap-and-transfer programmes creating and maintaining a lamprey population upstream of a dam is vital in attracting reliable recruitment of pre-spawning adults back to the base of the dam.

Advantages of upstream trap-and-transfer operations are that they provide temporary upstream passage. Research now indicates that careful consideration of stocking habitats and research to support stocking location is required (Félix et al. 2021). Studies post-stocking from tagged juvenile eels reveal low survival rates (up to 29%) and high site fidelity post-stocking (Camhi et al. 2021).

For lampreys relying on pheromone cues, upstream trap-and-transfer programmes can also be used to augment populations when monitoring indicates declining usage of a fishway by adult fish. For example, although lamprey passage structures for volitional movement of adult Pacific lamprey (*Entosphenus tridentatus*) have been present at hydro-electric dams in the Columbia River Basin, since 2015, a trap-and-transfer programme has been implemented at Wells Dam (9th dam upriver; CRITFC (2011). Over the past two decades it was thought that the pheromone cues produced by larval Pacific lamprey were no longer providing an attractive migration cue based on low numbers of adult lamprey passing Wells Dam and spawning in upstream tributaries. Monitoring of pheromones and eDNA is underway to assess the effectiveness of the trap-and-transfers (Lampman and Lumley (2020); Ralph Lampman, Yakama Nation Fisheries, pers. comm.).

Globally, there are limited options for providing fish passage at high-head dams, meaning there is a pressing need to advance the state-of-knowledge on trap-and-transfer programmes and their operations (Kock et al. 2021). Furthermore, trap-and-transfer programmes have evolved over time with technological advances, regulatory changes, improved biological knowledge and changing priorities that result in new goals and objectives (Kock et al. 2021). In Europe and USA, eel trap-and-transfer programmes are continuously being refined to set targets (e.g., stocking densities), and to achieve the outcomes expected (Matondo et al. 2020; Félix et al. 2021; Matondo et al. 2021; Newhard et al. 2021; Twardek et al. 2021). Establishing clear objectives and defined performance measures for trap-and transfer programmes in New Zealand is required to demonstrate their success.

Disadvantages of trap-and-transfer programmes are that a high degree of human intervention can have unintended consequences, including altering fish behaviour, influencing migration timing, and imposing selective pressure (such as size selectivity) that can collectively reduce life history variation and the resilience of the population (Lewis et al. 2022). Furthermore, upstream trap-and-transfer activities can compound existing and/or future issues with downstream fish passage. Where a series of dams exists, trap-and-transfer activities may only occur at the lowest barrier (Williams et al. 2022) and passage may not be provided for a portion of the population that continue to migrate upstream to the next dam.

Reviews of trap-and-transfer programmes for salmonids in the USA indicate there is no evidence for success and, given this information, new trap-and-transfer programmes should be carefully considered (Lusardi and Moyle 2017). Environmental factors such as water temperature and tidal height are known to influence eel trap-and-transfers for European eels (*Anguilla anguilla*; Piper et al. (2012)). Flow is also a factor that can influence the attraction efficiency of traps for lamprey (Rous et al. 2017; Tummers et al. 2018). However, in New Zealand, post-passage effects, fallback downstream over a dam after release, stress, low trap effectiveness, and migration delay have not been quantified.

While some trap-and-transfer programmes at high-head structures have been operational for decades, efforts to review and synthesise the effectiveness of these activities on a site-by-site basis or nationally is lacking. Little to no information exists for lamprey and surveys of eel population attributes such as size and age structure pre- and post trap-and-transfer activities indicate some improvements to the eel population occur, but this is generally hampered by poor baseline data from which to measure improvements and/or success.

Key considerations for upstream trap-and-transfer programmes include:

- Clearly identifying objectives and performance measures is imperative at the outset of trap-and-transfer activities (see Section 3).
- Site-specific information on the biology and ecology of the target species is required to inform an effective trap-and-transfer design. Understanding the size range of species to be captured is essential and trap designs need to account for differences in fish size attributable to site locations at different altitudes and distances inland.
- The siting of traps is important and capture efficiency can be highly influenced by trap location (Kock et al. 2021), although this has rarely been quantified in New Zealand. Size-specific differences in migration behaviours and route choice must be considered in siting traps (Kock et al. 2021). Choosing stocking sites so that post-stocking monitoring can be undertaken (Félix et al. 2021). Internationally, river sites with appropriate habitat conditions (Matondo et al. 2021; Newhard et al. 2021) and multiple sites within a river (Matondo et al. 2020; Félix et al. 2021) are being increasingly recommended for releases.
- Stocking density needs to be considered and stocking should only occur in sites with low densities (Degerman et al. 2019).
- Appropriate release sites with consideration to increasing the distance of release sites from the barrier to increase retention in upstream habitats (Twardek et al. 2021).
- Data to support identifying the appropriate stocking location includes understanding the habitat conditions that favour growth and survival following stocking (e.g., growth rates, size distribution), and examining ecologically relevant responses (e.g., sex differentiation etc.,) at different stocking rates/sites improve success following trap-and-transfer (Félix et al. 2021; Matondo et al. 2021). For some fish species (such as eels) the effects of stocking density on sexual differentiation must be considered (Newhard et al. 2021).

Ramp design

To enable fish entry into the trap, an inclined ramp with wetted climbing substratum is generally designed to create site- and species-specific solutions. The most common ramps utilised in trap-and-transfer programmes are termed eel ladders as they specifically target elvers or juvenile eels. Eel ladders are designed to enable elvers to use their climbing abilities to enter the trap. These structures are useful as elvers can preferentially employ or resort to climbing when it is not possible for them to traverse a fish pass using swimming alone (Padgett et al. 2020).

Lamprey-adapted fishways are also used globally as a means of improving passage of lampreys past hydro-electric dams. High passage success of lampreys has been found using purpose-built ramps that take advantage of their unique burst-attach swimming and climbing behaviour (Moser et al. 2011; Hume et al. 2020). Presently, only two lamprey species, the Pacific lamprey (*Entosphenus tridentatus*) and pouched lamprey are known to possess the ability to climb vertical surfaces using their oral disc with their body completely out of water. The unique climbing ability of lamprey has led to the successful development of Lamprey Passage Structures (LPS) to enable Pacific lamprey to migrate past hydro-electric dams on the Columbia River (Moser et al. (2011); Zobott et al. (2015); Frick et al. (2017); Figure 7-8). For effective passage of Pacific lamprey the LPS requires a completely smooth surface usually made from aluminium (Reinhardt et al. 2008; Moser et al. 2011; Frick et al. 2017). In contrast, for sea lamprey (*Petromyzon marinus*), which must pass instream obstacles using a burst-attach swimming mode, a studded ramp that takes advantage of their anguilliform swimming, has been shown to have high passage efficiency (Hume et al. 2020).

For both eel ladders and LPS, designs can be used as either volitional passage structures without the need for manual transfer, or the key components can be used to form the attractant ramp as part of a trap-and-transfer programme.

Figure 7-8: Lamprey passage structure (LPS) installed at Bonneville Dam, Columbia River, Oregon. Photo credit: Cindy Baker.

Several other New Zealand species, namely kōaro, giant and banded kōkopu, redfin bullies and torrentfish display a range of climbing abilities that enable bespoke ramps to be developed for capturing these species in trap-and-transfer programmes. A range of artificial substrate ramps have been tested for overcoming low-head vertical drops in New Zealand (see Section 5.5.2). However, there are limited quantitative data to support the design and placement of ramps at the base of dams for species other than elvers, lamprey and kōaro. The following section, therefore, provides guidelines on design features and identifying where knowledge gaps still exist in developing effective eel, lamprey and kōaro trap-and-transfer programmes.

Eel ladders

In New Zealand, there is, and has been, no effective volitional eel ladder functioning at any hydro-electric power scheme (Crow et al. 2020). While many eel ladders were tested, their failure to provide effective upstream passage can largely be attributed to poor design and/or maintenance of the structure and a lack of quantitative data to inform design criteria (Crow et al. 2020). Elver ladders were installed at several high-head structures in the 1980s, 1990s and 2000s (e.g., Matahina, Pātea, Wairere Falls, Arnold, Waitaki) and comprised a diversity of materials and substrates including plastic bristles and stones, with most passes less than 10 m in height (Jellyman and Arai 2016). Little monitoring or maintenance of eel ladders was undertaken, and even with subsequent retrofitting attempts all have ceased to function with issues such as rusting and lack of sufficient water supply (Crow et al. 2020). Furthermore, monitoring of eel passage efficacy was poor or non-existent and asset owners eventually resorted to upstream trap-and-transfer (Crow et al. 2020).

Gradient and length

For any fish pass, the length, gradient and substrate used on the structure are three key factors determining fish passage success (Baker and Boubée 2006; Baker 2014; Jellyman et al. 2016). Empirical data on elver passage over artificial passes are either for structures with a steep gradient, low head height and short length, or low gradient passes that can span longer distances. For example, to ensure passage of the American eel (*Anguilla rostrata*) past the Moses-Saunders Hydro-Electric Power Dam in the St. Lawrence River, Ontario, Canada, a 29.3 m high and 156.4 m long fishway was constructed at a gradient of 12° (Whitfield and Kolenosky 1978). The fishway contained willow cuttings stapled to the base to provide resting spaces and cover for migrating eels (Whitfield and Kolenosky 1978). During four years of monitoring, over 3 million eels passed over the dam. In contrast, for European eels (*A. anguilla*), Vowles et al. (2015) achieved up to 67% passage over a 1.25 m studded ramp with an angle of 11.3° and Watz et al. (2019) recorded 40% of elvers successfully passing a 1 m long studded ramp set at 30°. An examination of eel ramps, technical fishways and nature-like fishways in south-western Sweden found that low gradient nature-like fishways showed the greatest increase in eel numbers upstream of the dams (Tamario et al. 2019).

A New Zealand specific study by Jellyman et al. (2016) examined climbing success for shortfin elvers (< 155 mm) on three different ramp surface types (smooth plastic, sand and gravel, and Miradrain™) and three different slopes (30°, 50° and 70°). Results showed that shortfin elver climbing success decreased with increasing ramp slope and elver climbing ability was highest using the Miradrain™ surface. Importantly, the mean length of elvers successfully negotiating the ramps was longer than that of elvers that failed to climb and only larger elvers could climb the Miradrain™ surface as ramp slope increased. Therefore, the size-dependent climbing abilities of elvers are dependent on the characteristics of the features to be navigated. Overall, Jellyman et al. (2016) achieved >80% passage success of shortfin elvers over a 1.5 m long, 30° ramp lined with Miradrain™, with passage decreasing to below 20% as slope increased to 70°.

In general, the most effective gradients for an elver pass are between 15 and 30° (Knights and White 1998; Armstrong et al. 2010; Jellyman et al. 2016). Steeper ramp slopes are likely to raise energy requirements for climbing elvers, which, if coupled with a poor or suboptimal surface type, would likely lead to rapid fatigue and low or no successful passage (Jellyman et al. 2016; Tamario et al. 2019). Recommendations for gradient and length of an elver ladder for a trap-and-transfer programme are:

- At an angle of 30°, eel ladders should be no longer than 1.5 m.
- At an angle of 15°, using a recommended substrate, ladder lengths can extended given a resting pool is provided every 4–5 m (Armstrong et al. 2010; Environment Agency 2020).
- As per Figure 5-18, ramps should be tilted horizontally by 15° to ensure a wetted margin is present for climbing.
- There is no set width a ramp should be, but flow should be set to ensure a wetted margin exists based on the width of ramp chosen.

Substrate

Insights into the hydrodynamic implications of species-specific morphology can help in determining appropriate materials, in identifying proper patterns and distances between baffling elements for eel ladders (De Meyer et al. 2020). Myriad studies have tested multiple climbing substrata to develop ramp protypes for installation and subsequent field testing (Solomon and Beach 2004; Vowles et al. 2015; Jellyman et al. 2016; Watz et al. 2019). Of the substrates examined, the following are recommended for use or testing on New Zealand species:

Miradrain™ (Figure 7-9) has been proven as an effective substrate for climbing passage of elvers (Jellyman et al. 2016), redfin bully and weak-swimming fish such as īnanga (Baker and Boubée 2006). It is, therefore, the recommended substrate for trap-and-transfer programmes targeting elvers.

Figure 7-9: Miradrain™ substrate tested by Baker and Boubée (2006) and Jellyman et al. (2016).

Eel bristle substrates are a cost-effective, durable solution that can provide passage for a range of eel sizes when installed correctly. The high-density polyethylene (HDPE) backboards typically come in fixed lengths and widths and can be easily cut to fit any installation. Bristle spacing of either 20 mm or 30 mm are recommended (Environment Agency 2020). Two different bristle spacing’s on one board is recommended where a broader range of eel size classes are present.

In New Zealand, a nylon brush substrate (4 mm diameter heads at 18 and 28 mm spacing) has not been examined empirically for elver passage but has been tested by Baker and Boubée (2006) for passage of redfin bullies (Figure 7-10). Redfin bully are regarded as having the weakest climbing abilities but can climb along the shallow wetted margins of ramps, using a ‘breast-stroke’ movement, propelling themselves forward with their large pectoral fins and then adhering to the surface, again with the pectoral fins (Baker and Boubée 2006).

At a slope of 30°, the Miradrain™ ramp provided the highest passage success, with over 50% of bullies negotiating the ramp (Baker and Boubée 2006). As slope increased to 45°, passage success was less than 15% for all ramps, but bully surmounted the brush ramp more easily than the Miradrain™ ramp (Baker and Boubée 2006). The bristle substrate is thought to promote passage success of elvers through each bristle clump supporting anguilliform movement up the ramp. Should a bristle substrate be utilised at a trap-and-transfer operation targeting elvers, testing passage efficiency is warranted.

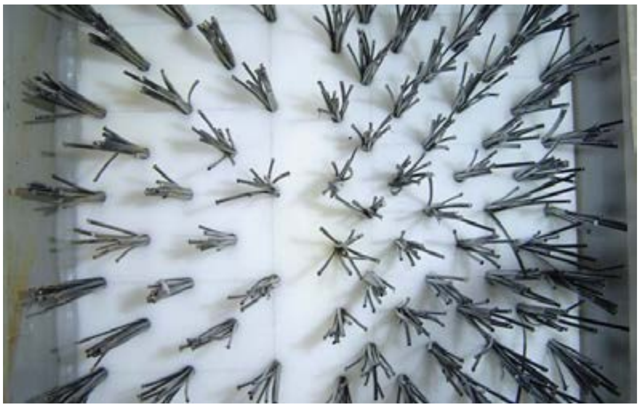


Figure 7-10: Nylon brush substrate (4 mm diameter heads at 18 and 28 mm spacing) tested by Baker and Boubée (2006).

Milieu Inc. Eel ladder is a synthetic substrate consisting of staggered cylinders on a flat base that has been used extensively in North America as eel ladder substrate (Milieu Inc.) for eels 150 mm to 750 mm (Figure 7-11). This substrate has not been tested with New Zealand eel species and empirical data on passage efficiency for the American eel is lacking. Theoretically, the Milieu Inc. Eel ladder could provide high passage success for juvenile eels >150 mm that tend to be present at dams further inland. However, laboratory or field testing with New Zealand eel species is necessary.

Figure 7-11: Milieu Inc. Eel ladder (for eels 150–750 mm) installed at the Moses-Saunders Dam. Photo credit: NY Power Authority.

A successful eel ladder will contain one of the substrates recommended above with a continual low flow that provides a wetted margin for climbing.

Lamprey Passage Structures (LPS)

LPS for Pacific lamprey started development in 2002 and by 2011 were successfully used to pass lamprey above hydro-electric dams in the Columbia River Basin (Reinhardt et al. 2008; Moser et al. 2011). The passage structures consist of a series of ramps and resting boxes (Figure 7-12). In development of the LPS, passage success was shown to be less influenced by ramp slope and flow and more so by substrate and entrance location (Moser et al. 2011). Combinations of slope and flow affected passage time but not passage success (Moser et al. 2011; Corbett et al. 2014). Pacific lamprey require a smooth surface for attachment and aluminium is usually used as the LPS substrate (Moser et al. 2011; Zobott et al. 2015; Lamprey Technical Workgroup 2022). The transition between the ramp and the resting box consists of a rounded crest with a radius of 8– 0 cm. This is because in areas of high velocities (above 1.0 m s^{-1}) where lamprey need to use burst-and-attach locomotion they can be hindered by squared corners (e.g., 90° angles) or tight radii corners because these sharp angles do not allow them to quickly re-attach to a flat surface after burst swimming (Lamprey Technical Workgroup 2022).

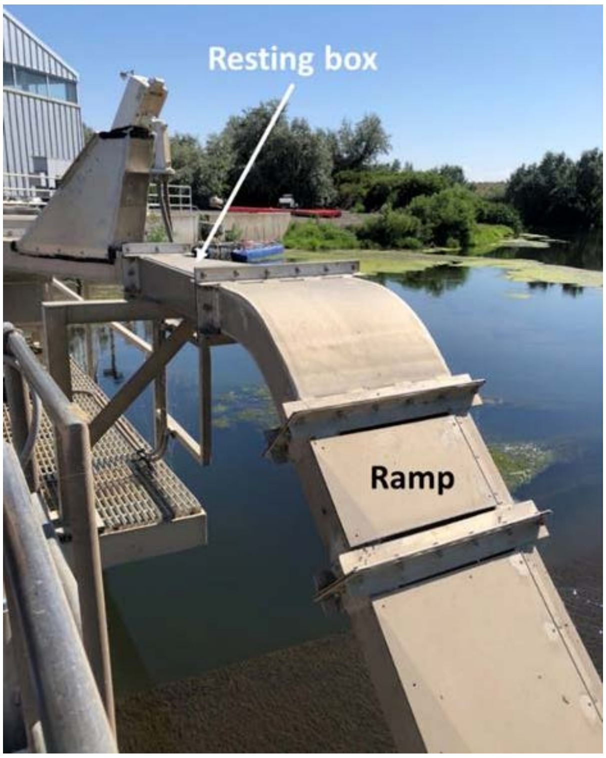


Figure 7-12: Example of a resting box in a LPS designed for Pacific lamprey passage at Three Miles Falls Dam, Umatilla River, Oregon. Photo credit: Cindy Baker.

To help develop an LPS for New Zealand pouched lamprey, NIWA carried out a laboratory study examining the most appropriate slope and substrate to promote pouched lamprey passage. Using a flow of 1 l s^{-1}, two slopes were tested (45° and 60°), alongside three substrates: stainless steel, textured high-density polyethylene (HDPE; Figure 7-13) and smooth HDPE. Results showed that a textured HDPE ramp set at 45° provided the highest passage success of lamprey (87.5%) in the quickest timeframe (NIWA unpublished data).

Another important feature of the ramp was the crest, where a flat transition enabled lamprey to pass the ramp using their burst-and-attach mode of locomotion. Without a flat transition to enable lamprey to effectively exit the ramp, rounded crests alone resulted in lamprey failing to leave the ramp and falling back to the bottom of the system.

In addition to LPS, tube fishways have been developed for Pacific lamprey passage. Goodman and Reid (2017) found a smooth 10 cm diameter tube (made of Acrylonitrile Butadiene Styrene) with a flow rate of 0.22 l s^{-1}, set at a 10% slope (5.7°) promoted 100% passage success of Pacific lamprey at a 1 m head differential. Tube fishways for lamprey have not been tested at high head structures but may represent a cost-effective option for attracting lamprey into traps where low gradient ramps can be utilised. Presently, the movement of New Zealand pouched lamprey within tube fishways has not been tested and a curved surface is not recommended without testing its attachment efficacy relative to a flat surface.

Figure 7-13: Lamprey climbing the textured HDPE ramp in the laboratory. Photo credit: Cindy Baker.

Based on laboratory findings and international studies with Pacific lamprey, the key criteria for developing a pouched lamprey trap-and-transfer system are:

- The textured HPDE substrate[17] (Figure 7-13) secured to a ramp base and set with a maximum gradient of 45° is recommended.
- The maximum length of the ramp should not exceed 3 m without resting areas provided.
- Ramps should not be tilted horizontally but remain flat to enable water to flow across the entire surface.
- Flow should be as low as possible but adequate to wet the full width of the ramp. Ramp widths of 0.6 m are recommended which will require a flow of approximately 1 l s^{-1}.
- Resting areas for lamprey are simply a horizontal, flat section in the ramp to enable the lamprey to rest and recover while attached to the substrate without the additional forces due to a gradient acting upon them (Figure 7-12). A rest area must be at least

[17] https://lep.net.nz/materials-plastics-selection-guide/hdpe-marine-board/

0.75 m in length to enable the entire body of all sized adults to be connected to the flat surface and effectively rest.

- An important factor when adding resting areas to the ramp is the transition between the sloped and flat surfaces. Rounded corners (8–10 cm radii or greater), are recommended when ramps transition into, and out of, resting areas.
- A similar low radius ramp crest (8–10 cm or greater) that transitions to a flat section of ramp prior to the lamprey exiting the ramp and entering the trap is an extremely important design feature (Figure 7-14). The length of the flat crest section should be a minimum of 0.25 m to enable around half of the lamprey's body length to be supported during movement over the crest and then into the trap.
- To ensure lamprey cannot exit the trap once inside, the ramp flow needs to be separate from the trap holding area (Figure 7-14). A single inflow of water into the trap to overtop and flow down the ramp should not be used (Figure 7-14). A drop from the ramp of at least 0.5 m is recommended to prevent lamprey jumping back onto the ramp. The flow for the ramp needs to be pumped separately to the ramp crest, with a secondary water supply pumped into the holding area of the trap to circulate and drain to waste. This ensures the water quality and oxygen is maintained as lamprey numbers increase in the trap and that the water temperature does not rise within the trap.

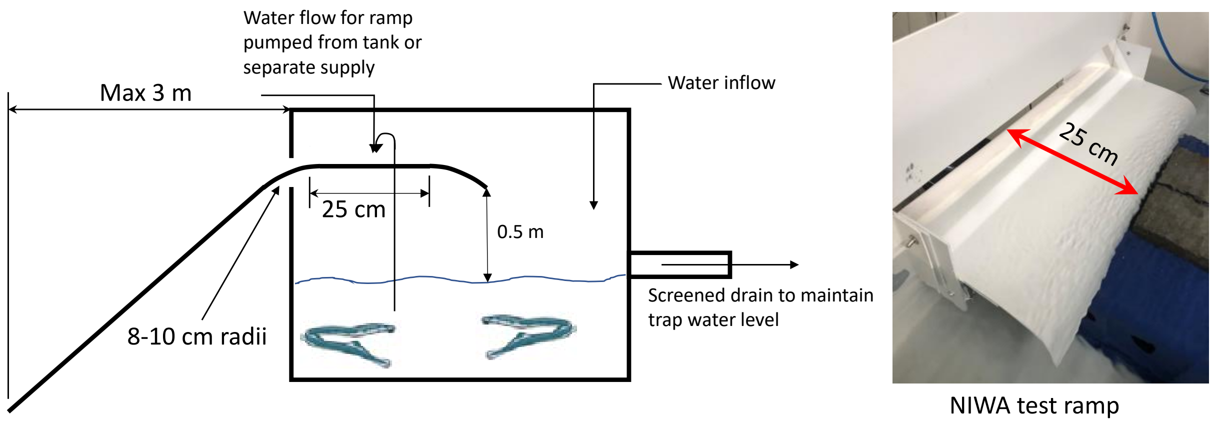


Figure 7-14: Schematic (side view) of key elements of a lamprey trap to prevent lamprey exiting the trap and an example of a flat transition post-ramp crest to facilitate lamprey entry into the trap.

Kōaro ramps

To provide advice on ramps suitable to allow trap-and-transfer of kōaro, NIWA carried out a laboratory study examining the most appropriate slope and substrate to promote passage of juvenile kōaro. Using a flow of 0.018 l s^{-1} (essentially a trickle to wet the ramp), three slopes were tested (45°, 65°, and 85°), alongside two substrates: textured high-density polyethylene (HDPE; Figure 7-13), and smooth HDPE. Results showed that a textured HDPE ramp set at 45° provided the highest passage success of kōaro (NIWA unpublished data).

Figure 7-15: Kōaro climbing a smooth HDPE ramp in the laboratory. Photo credit: Eleanor Gee.

Based on laboratory findings, the key criteria for developing a kōaro trap-and-transfer system are:

- The textured HPDE substrate[15] (Figure 7-13) secured to a ramp base and set with a maximum gradient of 45° is recommended.
- Ramp length can be up to 1.5 m. Longer lengths are likely to be passable but should be tested first.
- As per Figure 5-18, ramps should be tilted horizontally by 15° to ensure a wetted margin is present for climbing.
- Flow should be as low as possible but adequate to wet the ramp. In the laboratory trials the flow was essentially a trickle. A corrugated profile at the top of the ramp could be used to provide several areas for a very small flow to trickle onto a wide ramp.
- There is no set width a ramp should be; however, flow should be set to ensure a wetted margin exists based on the width of ramp chosen.
- To minimise the possibility that kōaro exit the trap once inside, like lamprey, the ramp flow needs to be separate from the trap holding area (Figure 7-14), and the holding tank should have a lid. A single inflow of water into the trap to overtop and flow down

the ramp should not be used (Figure 7-14). The flow for the ramp needs to be pumped separately to the ramp crest, with a secondary water supply pumped into the holding area of the trap to circulate and drain to waste. This ensures the water quality and oxygen is maintained as kōaro numbers increase in the trap and that the water temperature does not rise within the trap.

Entrance location and attraction flow

The entrance location and attraction efficiency of the ramp are two of the most important factors in an effective trap-and-transfer programme. The downstream entrance should be in an area where the target species will congregate, which is generally defined as the most upstream point of migration when reaching an impassable barrier (see Section 5.5.3 for further details). For elvers, this will usually be in an area where there is a low velocity flow away from the obstruction, not an area where velocity is high (Armstrong et al. 2010; Watz et al. 2019). For Pacific lamprey, adults are attracted by water volume and velocity, where the fishway flow must be sufficient and have a sufficient water velocity to provide upstream migration cues at the entrance (Keefer et al. 2010; Johnson et al. 2012; Kirk et al. 2015). Attraction of adult pouched lamprey to fishways or ramps has not been empirically quantified but could be like that recorded for Pacific lamprey. At the base of impediments, pouched lamprey have been observed to congregate in a low velocity zone, adjacent to areas of high water velocities, with climbing attempted in the high velocity water (> 1 m s^{-1}; NIWA unpublished observations). There is currently no empirical data on the behaviour and attraction to fishways or trap-and-transfer facilities by kōaro.

Attraction flows can be added to the ramp entrance at trap-and-transfer operations to increase attraction efficiency and entrance of target species. For example, at Karāpiro and Matahina Dams, elvers have been found to aggregate in the warm discharge associated with transformer cooling water outlets (Crow et al. 2020). This is not unexpected as eels are the most temperature tolerant of all native fish species, and Richardson et al. (1994) recorded the temperature preferences of shortfin and longfin elvers at 26.9°C and 24.4°C, respectively. Studies on the effects of plunging and streaming flow at eels ramps (situated in an inter-tidal context) found eel passage was two-fold higher when plunging flows were provided (Piper et al. 2012). At Bonneville Dam, the flow adjacent to the LPS situated in the Bradford Island auxiliary water supply channel was found to attract adult Pacific lamprey to the LPS entrance (Moser et al. (2011); see Figure 7-8). However, the effectiveness of augmented flows can be contingent upon fishway flow. For example, background turbulence levels can cause both an attraction and avoidance behaviour in Pacific lamprey relative to turbulence in the fishway (Kirk et al. 2015). Pheromone cues can also enhance the attractiveness of a water supply to lamprey (Johnson et al. 2015) and supplying the trap and ramp with water sourced from a waterbody where a lamprey population has been confirmed could enhance efficiency of adult entry.

In general, manipulating hydrodynamic conditions can enhance passage efficiency and attraction to ramp entrances (Drouineau et al. 2015). Unfortunately, anticipating the behaviour of elvers, lamprey and kōaro in response to a specific flow pattern is difficult because swim path selection and response to turbulence and other hydrodynamic conditions varies considerably among species and sites (Williams et al. 2012; Heneka et al. 2021). Size-specific differences in migration behaviours and route choice must also be considered in siting trap entrances (Kock et al. 2021). As such, there is no guarantee that elvers, lamprey and juvenile kōaro will be attracted to the same hydrodynamic conditions, as well as to the proposed location of the fish pass entrance.

Volitional entry into fishways continues to be a challenge worldwide (Zielinski et al. 2020) and presently, there is too little evidence to provide guidelines on attraction flows for New Zealand operations. The entrance location for a trap should start at the upstream point of migration for the target species (see Figure 5-12) and will need to be guided by monitoring the efficiency of entry.

7.3.4 Fish lifts and locks

Mechanical-type fishways are another non-volitional method used to raise fish over high-head dams (suited to barriers between 5–14 m), but they can also be used at low-head structures (e.g., 3 m; (O'Connor et al. 2017b). Various mechanical-type designs such as fish lifts and locks have been designed worldwide, both automatic and manual systems. While mechanical-type fishways can pass a wide range of fish species, they were originally designed in the Northern Hemisphere, for salmonids and weak-swimming species, being less suitable for bottom- dwelling, and small-bodied fish. One of the main challenges for each design is efficiently collecting and delivering the fish to the lift, lock, or pump intake. Unlike other fishways, fish locks and lifts operate cyclically to trap-and-transfer fish past considerable migration barriers. Consequently, it is essential to schedule an operation cycle and maintenance operation seasonally or continuously according to the ecological importance of migratory fish species. These fishways require considerable maintenance to ensure effective operation and are often subject to mechanical failure leading to poor performance.

Fish lifts

Fish lifts operate like an elevator, using a substrate ramp to encourage fish to enter a holding tank, which is then lifted to the head of the dam or above head pond level. Depending on the design, the lift empties into the head pond, the fish swim out of their own accord, or the fish may be removed manually by an operator (Figure 7-16). Experience in New Zealand and internationally has shown that fish lifts can be an effective method for fish passage at high-head dams. Trap-and-transfer operations that are successfully operating throughout New Zealand are based on the design of fish lifts (e.g., see Section 7.3.3). Some international studies have found the efficiency of fish lifts to be generally poor (the number of fish exiting the fishway as a proportion of those entering) for salmonids and non-salmonids (Noonan et al. 2012). A main problem associated with fish lift efficiency is related to poor attraction into fish lift structures (Larinier et al. 2005; Croze et al. 2008). This is related to the siting of the fishway entrance and the magnitude of the attraction flow. Consequently, auxiliary water release is often used to provide attraction flows that are competitive in comparison to other project flows. Fish lift attraction flows will vary according to target species but at least 1–5% of the project flow is recommended (Larinier and Travade 2002). However, flows up to 10–20% of the turbine discharge may be necessary if functional fish passage using a lift is desired (Larinier et al. 2005). One of the main limitations of fish lifts is that due to their mechanical nature, they require regular maintenance to ensure they remain operational and can, therefore, be expensive to implement.

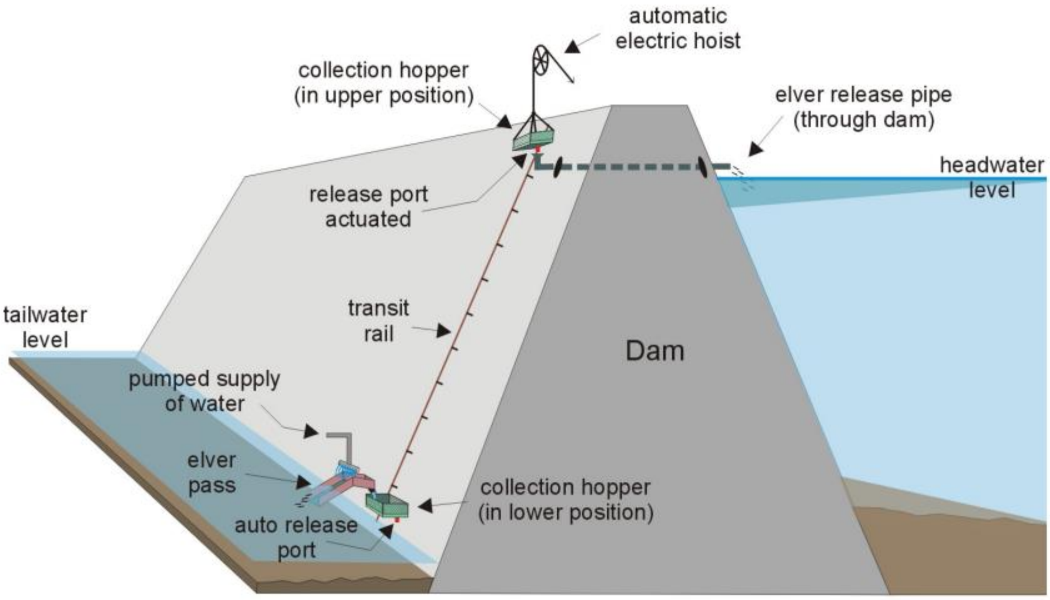


Figure 7-16: Conceptual diagram of a fish lift targeting eels (from Solomon and Beach 2004).

Fish locks

Fish locks operate by attracting fish through an entrance like that of a pool-type fishway, but instead of swimming up a channel the fish accumulate in a holding area at the base of the lock. This holding area is then sealed and filled with water to reach a level equal to the water level upstream of the barrier. Fish are then able to swim out of the lock (Thorncraft and Harris (2000); Figure 7-17). No fish locks are currently in use in New Zealand, but a number have been used successfully in Australia. Because of the limited technical guidelines on the design of fish locks for New Zealand, research would be required to design this structure for passage of native New Zealand species. Key features critical to success include entrance location, attraction flows, and sizing appropriate to the species and biomass of fish expected to be moved.

General disadvantages of lift and lock methods include the high construction and maintenance costs and the requirement of considerable engineering expertise (Boubée et al. 2000). While a large range of fish species may be moved upstream using these methods, they are not suitable for all species.

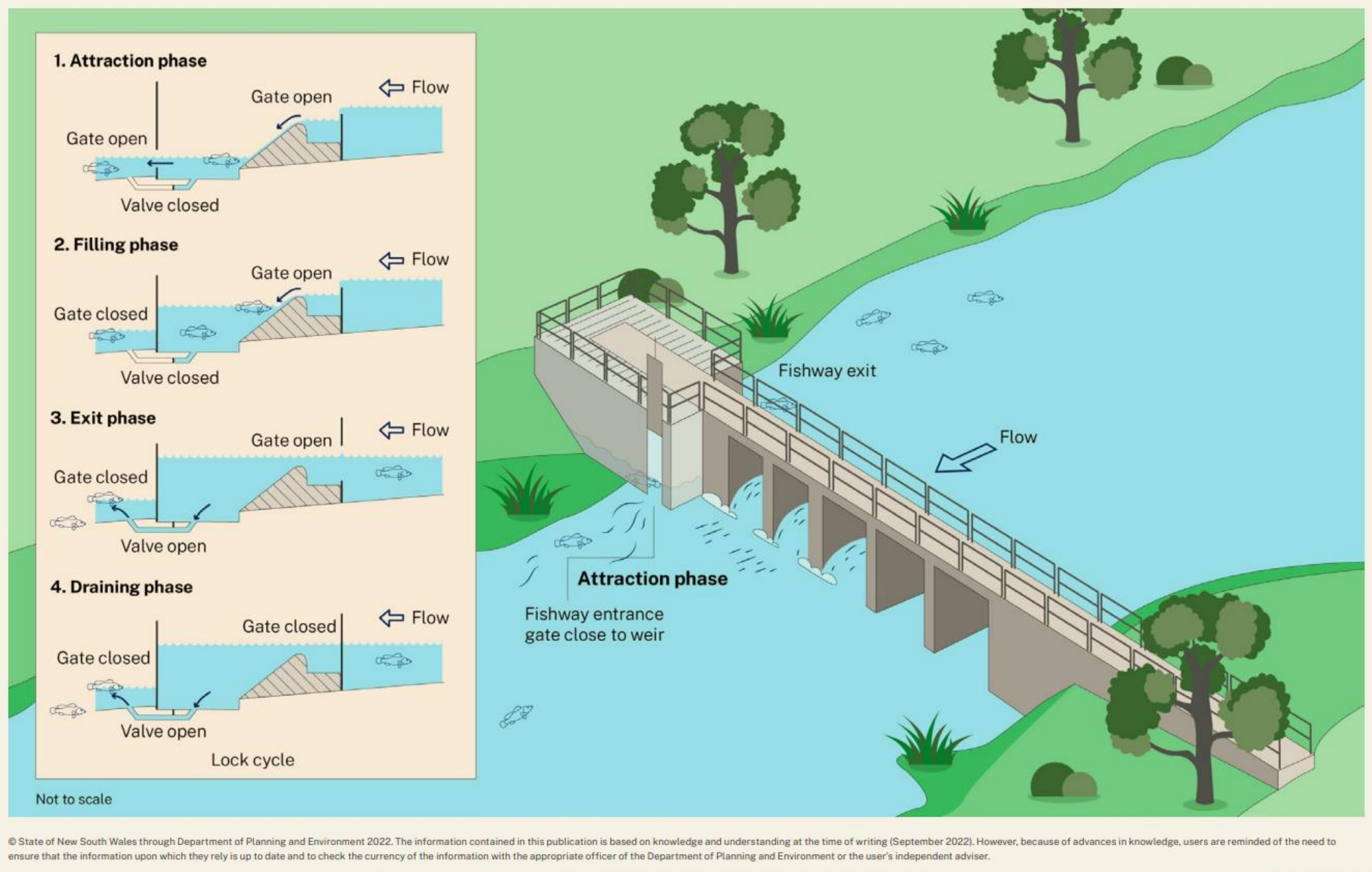


Figure 7-17: Conceptual diagram of Fish lock mechanical fishway . Source: Department of Planning and Environment, NSW.

7.3.5 Technical Fishways

Technical fishways are bypass structures constructed in engineered channels, that are commonly used internationally to provide fish passage past structures when passage cannot be restored through other means. Technical fishways are most effective at facilitating fish passage past low- to medium-head obstructions and are typically dependent on relatively strict design criteria to provide conditions suitable for target fish species. In New Zealand, few technical fishways have been installed. The reason for this is mainly due to the design guidelines specific to New Zealand fish species only recently having become available. Because most technical fishway designs originated from efforts to promote the passage of salmonid species, there is a general lack of evidence supporting fishway design for species in the temperate south (Stuart and Mallen-Cooper 1999; Link and Habit 2014; Wilkes et al. 2018a). However, more recently these designs have been adapted and tested to suit a much wider range of fish species, including weaker-swimming fishes (e.g., vertical slot-fishway designs suitable for īnanga).

Common technical fishway types used internationally and described in this section include:

- **Pool-type fishways**: Pool and Weir, Vertical slot fishways, Trapezoidal fishways and Cone fishways
- **Baffle-type fishways**: Denil fishways

Pool-type fishways

Vertical slot fishways

Vertical slot fishways (VSFW) are generally installed at low- to medium-sized dams up to 6 m high. The basic design of a VSFW is a rectangular concrete channel structure that is partitioned by baffles into individual resting pools (Figure 7-18). The vertical slot runs the entire depth of the baffle and angles the water jets across the pool to the opposite side. This dissipates hydraulic energy within each pool, allowing fish to swim from pool to pool through each slot (Thorncraft and Harris 2000). The vertical drop between each pool, the pool dimensions, and the slot width connecting each pool determines the turbulence and maximum water velocity parameters of the VSFW. The appropriate dimensions must be dictated by the size, biomass, and species of fish for which passage is to be provided. This in turn influences the gradient, length, and cost of the fishway. Because the slot width specifically determines the maximum size of fish that can use the fishway, this means that the slot needs to be large enough for the largest species to physically fit through (O'Connor et al. 2017b).

Three main sub-types of VSFW are generally installed, based on the shape/s of the slot in the baffles (O'Connor et al. 2017b):

- Standard VSFW (long rectangular shaped slot, insert in Figure 7-18).
- Keyhole VSFW have variable slot width and shape, with a wider slot at the bottom. This allows a single fishway to pass small- and large-bodied fish by reducing turbulence and without increasing pool size (insert in Figure 7-18).
- Dual/Multi VSFW have more than one type of slot, e.g., a combination of standard vertical and keyhole (Figure 7-19).

In addition, several new variations in VSFW design include:

- E-nature VSFW which can have a >30% lower energy dissipation rate and fish pass discharge, and approximately 20% lower water velocities in pools and slots compared to a standard VSFW (Mader et al. 2020).
- Multiple or 'paired' hydraulically distinct VSFW (also called Sister VSFW) have been designed for sites with variable head differential and can operate optimally during different but complementary hydrological conditions (Bice et al. 2017).

VSFW are uncommon in New Zealand, and where they have been built, little monitoring has been conducted to determine their effectiveness (e.g., Mararoa Weir, Manapōuri Power Scheme, and Lake Taharoa). Although VSFW were historically considered inefficient and inappropriate for passage of New Zealand native fish species, recent work in Australia has demonstrated that low-head, low gradient VSFW can be successfully passed by some native New Zealand species, including īnanga (weak swimming), lamprey and shortfin eel at varying life stages (Morgan and Beatty 2006; O'Connor et al. 2017b). For example, research on adult lamprey (*G. australis*) passage through VSFWs has shown they can effectively use sequential VSFWs in the Murray River Australia and migrate up to 800 km upstream achieving a maximum migration rate of 40 km per day (Bice et al. 2019). Lamprey passage efficiency was 71–100% and 78–100% for the VSFW with slopes of 0.043 m/m and 0.031 m/m, respectively. Lamprey ascent rates varied with fishway slope and length, with longer and steeper fishways causing longer ascent times (Bice et al. 2019). Investigations of Pacific lamprey passage through vertical slot fishways indicated a surface for continuous attachment is the key factor in determining passage success through high velocity constrictions (Lamprey Technical Workgroup 2022). In areas of high velocities where lamprey need to use burst and attach locomotion, they can be hindered by squared corners (e.g., 90° angles) or tight radii corners because these sharp angles do not allow them to quickly re-attach to a flat surface after burst swimming. Therefore, rounded corners (8–10 cm radii or greater) at the orifice walls are advised on the up- and down-stream side of each wall.

While VSFWs are not common in New Zealand, design criteria for VSFW developed for small-bodied Australian species (which include īnanga) are an appropriate starting point for restoring fish passage for New Zealand species (Table 7-2; O'Connor et al. (2017b)). The recommendations below for VSFW target small-bodied fish (25–150 mm long), as these are the weakest swimmers. Because each site and fish community are unique, it is suggested that the individual criteria are refined as part of a collaborative process for any new or retrofitted fishways. Further specifications for medium to large-bodied fish are found in O'Connor et al. (2017b).

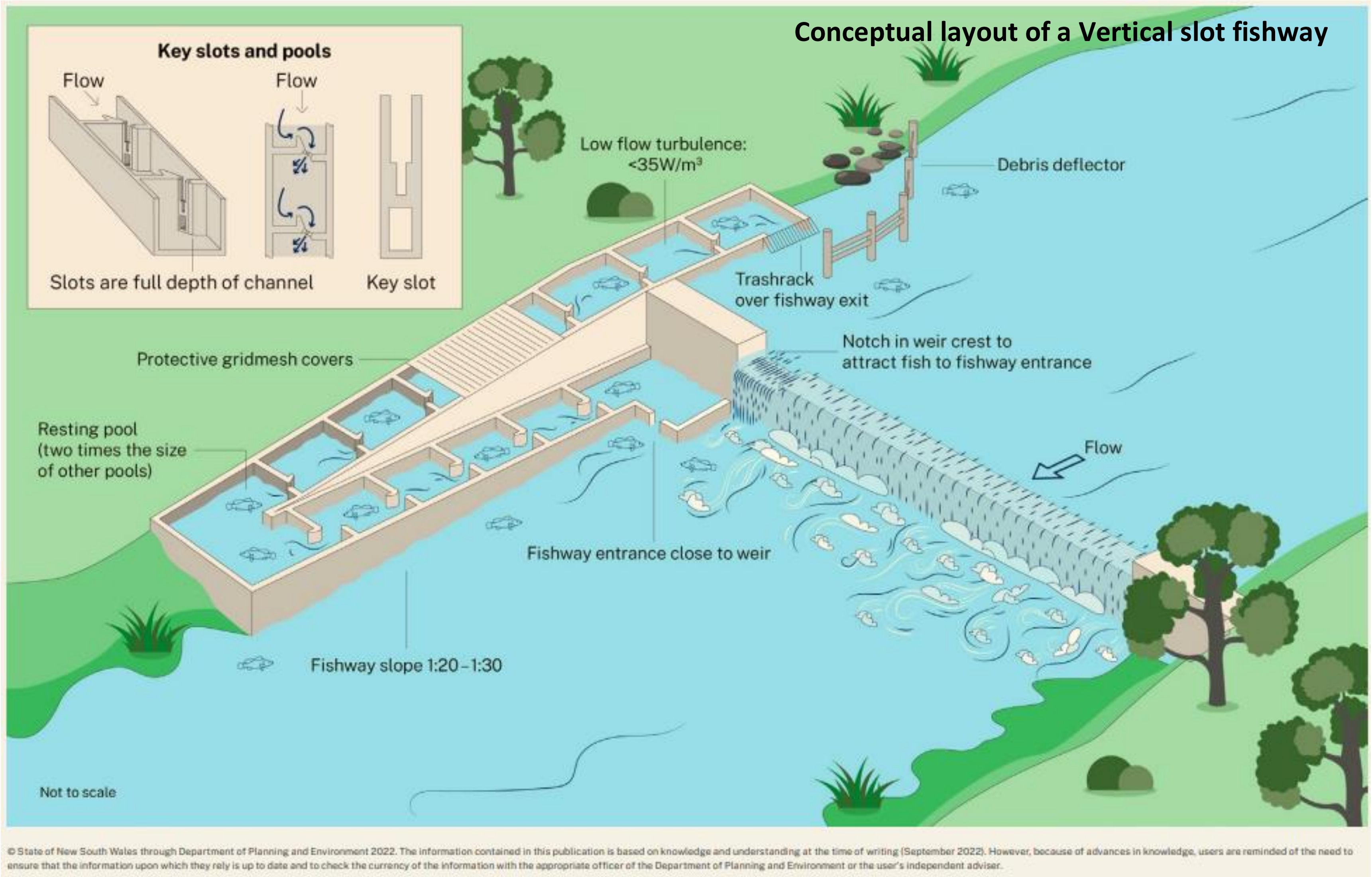


Figure 7-18: Conceptual diagram of a vertical slot fishway (from Department of Planning and Environment, NSW 2022).

Figure 7-19: Examples of different vertical slot fishway types. A = regular vertical slot fishway (Photo credit Ivor Stuart), B = dual/ multi slot fishway (Photo credit: Tim Marsden), C = E-nature vertical slot fishway (Reproduced from Mader et al. (2020)).

Table 7-2: Recommended physical and hydraulic standards for small-bodied fish at vertical slot fishways (up to 6 m head differential; O'Connor et al. (2017b).

Specifications	Design guidelines
Fishway operating range and differential head	The range of flows and differential head over which the fishway operates is a site-specific decision, but the standard criterion of fishway operation up to and including a 1-in-5-year flood is a baseline requirement.
Pool volume	A pool volume of at least 1.5 m^3 or 825 l is recommended (1.5 m long × 1 m wide) to allow for dissipation of energy to maintain acceptable turbulence levels; however, this is highly dependent on the slot width and head drop between pools, plus the size and biomass of fish expected to use the fishway.
Minimum depth	The minimum depth recommended for small-bodied fish is 0.4–0.5 m. For fish greater than 100 mm total length, minimum depth increases to 0.8 m.
Slope	A slope of >1:30 (vertical: horizontal) is recommended for the passage of small-bodied species, but there is scope to steepen the fishway where head loss and turbulence are low.
Slot width	A slot width of 0.15 m is appropriate in many situations, but narrower or wider slots (or keyhole slots) may be used where appropriate for the fish species and pool hydraulics. A general rule of thumb is slot width is ⅓ of the length of the longest fish. For 'bendy' fish such as eels, slot width is ⅓ of half the length of the longest eel requiring passage.
Head differential	The head loss between pools in vertical slot fishways should be kept low (0.05–0.1 m).
Hydraulics	Water velocity should be <1.22 m s^{-1} at the vena contracta[18], and <0.15 m s^{-1} at fishway exit channel. Turbulence should be <25 W m^{-3} for the passage of small-bodied fish.

VSFW's self-adjusting design, and ability to maintain constant velocity and turbulence levels throughout the channel at varying flows, means that VSFW can operate to allow fish passage over a wide range of head- and tailwater levels. Disadvantages of VSFW are that these can be more costly to construct and retrofit.

Section 7.3.1 outlines five key criteria to consider when designing the physical parameters of a fish way, such as a VSFW. In particular, the entrance location plays a critical role in determining the structures' ability to attract fish. The ease of access to the fishway entrance greatly influences whether fish can readily locate and enter the fishway for upstream migration. To efficiently attract fish to the entrance of a fishway, it must be located at the 'upstream limit of migration' for upstream migrants, which is confirmed by flow vectors, water velocity, and zones of turbulent water. Furthermore, the placement of the fishway entrance should be such that it avoids creating flow patterns that deviate more than 90° from the centreline of the stream or river. This helps prevent the formation of recirculation or eddies that could hinder fish passage (O'Connor et al. 2017). In addition, it is essential to ensure that the entrance location effectively attracts fish at the full operational flow range.

[18] A point in a fluid stream, just downstream from a restriction like an orifice, where the stream's cross-sectional area is at its smallest and the fluid velocity is at its highest. This phenomenon is significant in fish passage as it represents the highest velocity point in the flow, which could impede fish movement if the velocity is too high.

Further research is required to confirm the effectiveness of VSFW designs for some New Zealand fish species. As such, any new installation should include outcome monitoring to determine its effectiveness for target species.

Cone fishways

Cone fishways are a relatively new technical fishway design that have been widely adopted in Australia and increasingly in South-East Asia (Baumgartner et al. 2020). Cone fishways consist of a series of prefabricated cone-shaped concrete baffles installed across a concrete channel with a fixed crest level (O'Connor et al. (2017b); Figure 7-20). The design principal of cone fishways is similar to rock ramp fishways, providing multiple ascent pathways, instead of a single-slot design (Baumgartner et al. (2020); Figure 7-21).

Cone fishways typically have a relatively gentle slope of 1:20–1:30 (vertical: horizontal) and are most suitable for sites with a narrow headwater range (< 0.4 m; Baumgartner et al. (2020)) and relatively low-head differential (<1.5 m, O'Connor et al. (2015a)). However, successful trials have been conducted at head differentials up to 3 m (Stuart and Marsden 2021). There are two main sub-types of cone fishways: low cone fishways and high/low cone fishways, which refer to the height of the cones in the channel (O'Connor et al. (2015a); Figure 7-20).

The generic design elements of the cone fishway include a 2.4 m wide, 1.0 m deep channel, with pool volume between 2.52–3.60 m^3, depending on depth. The pre-cast cones are 0.2 m thick with a trapezoidal cross-section, and each row of baffles are set 1.5 m apart creating a 0.08 m head drop between each row (Stuart and Marsden (2021); Figure 7-20). This gives a maximum water velocity of 1.25 $m\ s^{-1}$ and a theoretically calculated average volumetric dissipated power (i.e., average pool turbulence) of 12 $W\ m^{-3}$ (C_d = 0.70). At each turn of the fishway channel, a larger pool twice the standard pool volume should be installed. Certain elements of cone fishways such as cone design, alternating the design through the fishway, pool depth and width, and head differential between baffles, can be modified to improve hydraulic performance and accommodate specific site or species constraints (Thomson and Redenbach 2022). However, to date, only the generic design elements of cone fishways have been tested on small bodied species.

Cone fishways are designed to enable bi-directional fish passage and are particularly appropriate for the passage of small-bodied species including galaxiids (īnanga) and Australian longfin eels and elver (Stuart and Marsden 2021). However, their effectiveness for passage of larger species or different life stages exceeding 300 mm is currently unknown (Stuart and Marsden 2021).

Some advantages of cone fishways include their simple design, low maintenance requirements, the possibility of pre-fabricated baffles to reduce overall construction costs, lower average turbulence compared to a vertical slot fishway, and the fixed crest that prevents complete drainage from a headwater pool while allowing safe human access and egress to meet safety standards (Baumgartner et al. 2020). To date, the effectiveness of cone fishways has only been evaluated for small-bodied diadromous species in Australia and South-East Asia. The results for small-bodied Australian species suggest this may be an effective solution for New Zealand species at sites with lower head differentials.

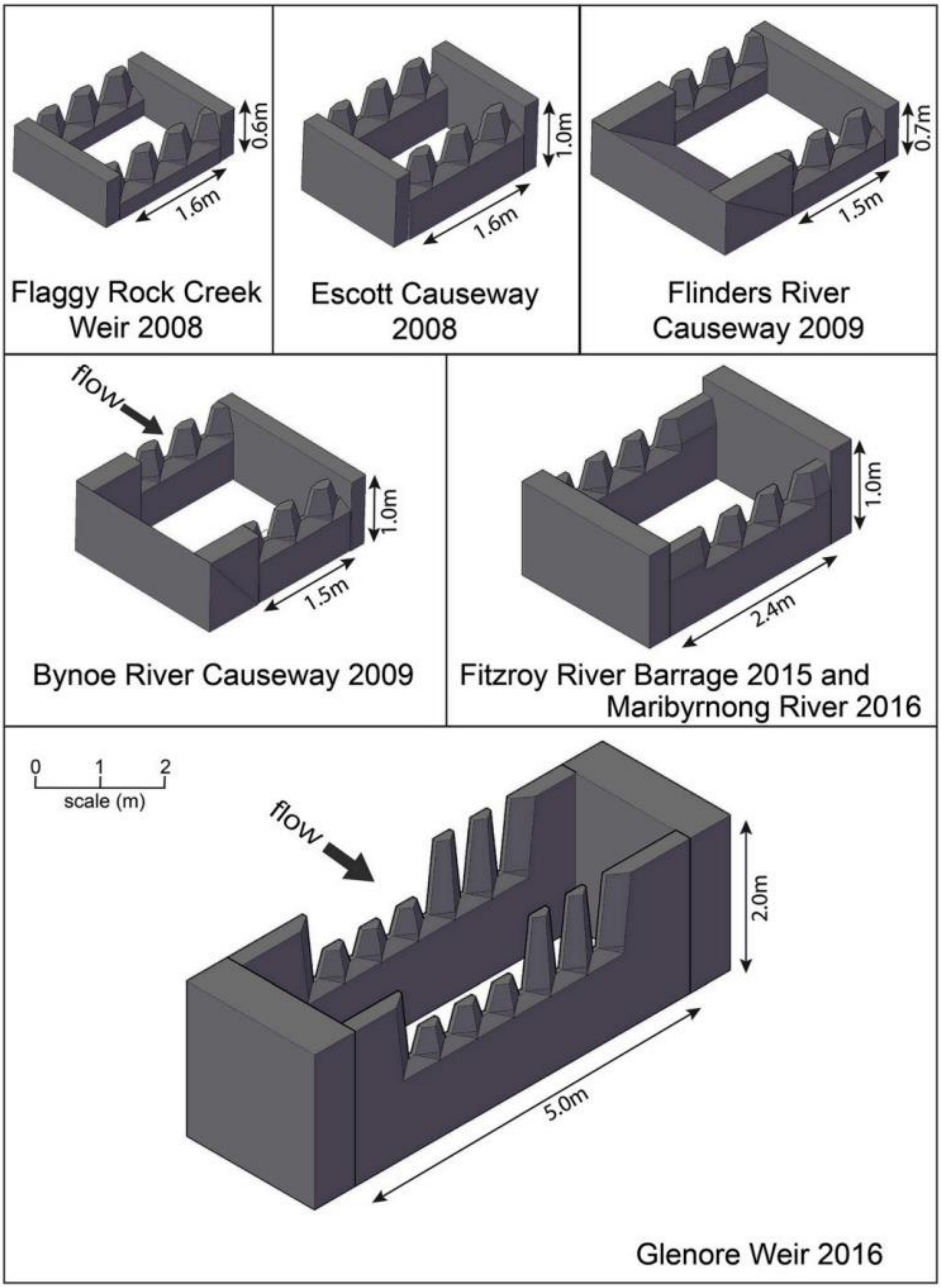


Figure 7-20: Scale-drawings showing the design evolution of the pre-cast cone fishway. Source: Stuart and Marsden (2021).

Figure 7-21: Examples of cone fishways. Panel a) and b) shows a high/low cone bypass (at Glenore weir fishway) at high and low flows. Panel c) shows a low cone bypass at the Fitzroy River Australia and d) is a tailwater stabilisation pool which improves hydraulics. Source: Stuart and Marsden (2021).

Trapezoidal fishways

Trapezoidal fishways are an emerging pool-type design, with a straight channel divided into pools with weirs (Figure 7-22). Many trapezoidal fishway designs are modified VSFW. The main difference of the trapezoidal fishway compared to the standard design of a VSFW remains in the separation of the pools into two zones: the migration corridor and the energy dissipation zone. Trapezoidal fishways are designed to ensure a low turbulence, vortex free area that improves upstream migration performance (O'Connor et al. 2015a). For the passage of small-bodied fish species through trapezoidal fishways, a slope of 1:30 (vertical: horizontal) is commonly employed. In cases where an additional fish passage structure is incorporated for this size range (e.g., a fish lock), the accompanying fishway is typically constructed with a steeper slope of 1:18 (O'Connor et al. 2015a).

Trapezoidal fishways are relatively untested in Australia (O'Connor et al. 2015a) and have not been tested on species relevant to New Zealand. Anecdotal evidence from Australia suggests that current designs are less effective for passing native weaker-swimming, small-bodied fishes compared to VSFW or rock ramp fishways (Tim Marsden, pers. com.). This design is currently not recommended and requires more research to determine the applicability for New Zealand fish species.

Figure 7-22: Trapezoidal fishway, Wyong River, New South Wales. Photo credit: Tim Marsden.

Pool and weir

Pool and weir fishway designs are the most common fishway type applied worldwide (Larinier 2002). However, they are not commonly used in New Zealand. These fishways originated in the Northern Hemisphere where they were largely designed to facilitate the migration of salmonid species. Pool and weir designs are installed at low- to medium-head applications (<3–4 m) and consist of a series of interconnected pools separated by low weirs. Historically, there were two types of designs:

- the submerged orifice design favouring only bottom dwelling fish, and
- the weir type design favouring jumping species such as salmon.

Where pool and weir fishways have been installed in Australia, many have failed or are in various states of disrepair as none have successfully provided fish passage for native species. Pool and weir fishways have also been deemed ineffective as they are not capable of maintaining designed water velocities because of fluctuating headwater and tailwater levels (Thorncraft and Harris 2000).

Traditional pool and weir fishways are not recommended for use in New Zealand. More recent variations on the pool and weir design, such as vertical slot fishways (discussed above), have proven more effective for New Zealand and Australian native fish species (O'Connor et al. 2017b).

Baffle-type fishways

Denil fishways

Denil fishways are open channels containing a series of symmetrical upstream-sloping ‘U’-shaped baffles without intervening pools. The ‘U’ -shaped baffles redirect the flow at the base of the channel creating a low velocity zone that fish use to ascend (Thorncraft and Harris (2000); Figure 7-23). These fishway designs offer the shortest upstream route around vertical barriers and are often installed in relatively steep slopes (e.g., 1:12 to 1:8 vertical: horizontal, Larinier (2002). Their use is generally reduced at high-head dams because there are no resting pools and fish must pass the fishway in only one attempt. Consequently, Denil fishways are only recommended for strongly swimming migratory fish, e.g., adult salmonids and trout. Due to the design’s high water velocity, Denil fishways may be quite selective and not functional for juvenile and weak-swimming species (Nielsen and Szabo-Meszaros 2022).

No Denil fishway designs have been installed in New Zealand or tested on fish species native to New Zealand. As such, there are currently no design criteria for this fishway type suitable for native fishes. In Australia, research has indicated the potential of Denil fishways for some native fish species. Because Denil fishways tend to favour fish greater than 40 to 60 mm in length (O'Connor et al. 2017b), and the passage of bottom- and midwater-dwelling fish species, poor passage has been reported for surface-dwelling species such as mullet (Baumgartner 2006; Mallen-Cooper and Stuart 2007). The main advantage of Denil fishways is that they can be built on steeper slopes than pool-type fishways such as the vertical slot designs (O'Connor et al. 2015a; Marsden et al. 2016), but they tend to be far more selective in terms of what species and life stages are able to pass.

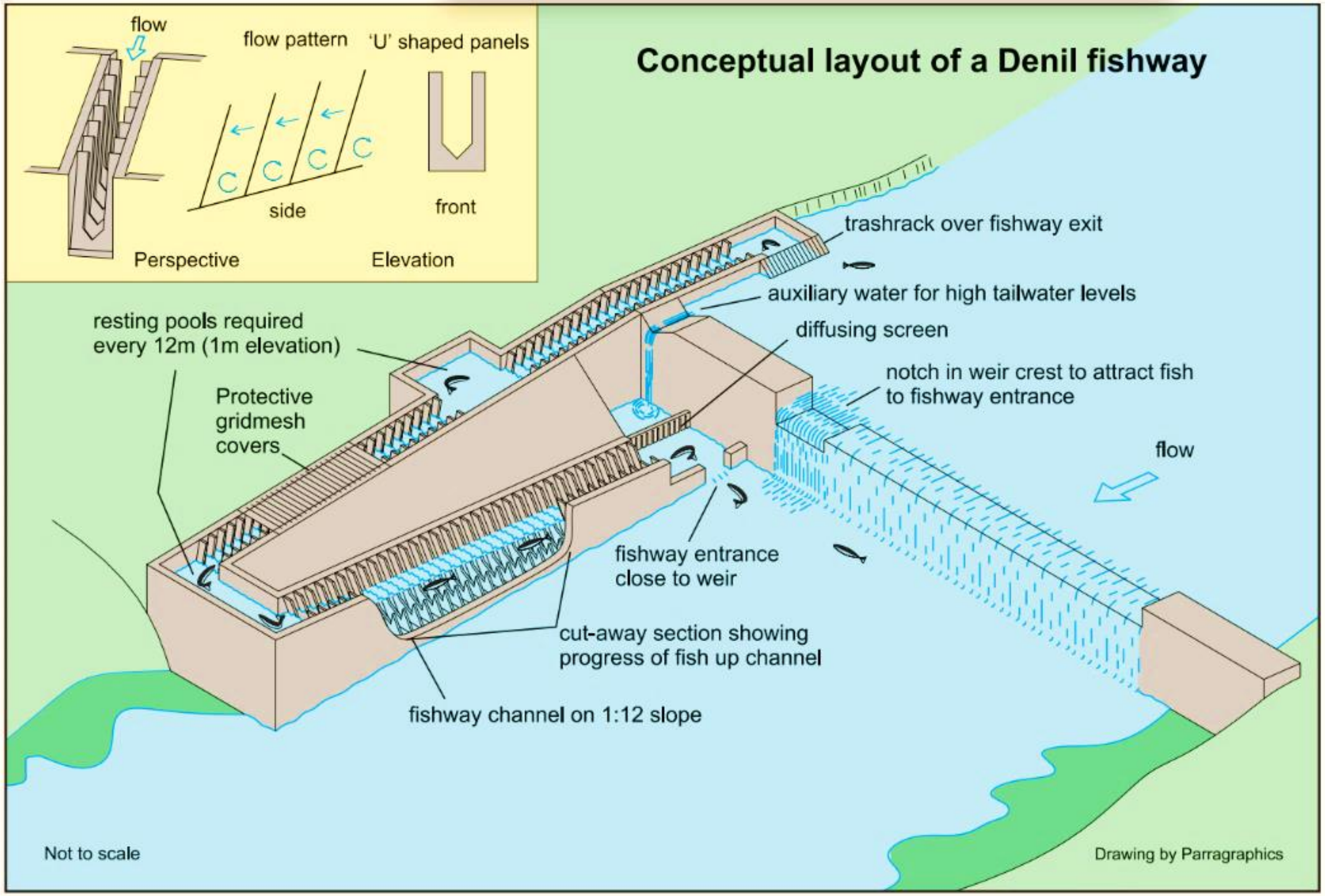


Figure 7-23: Conceptual diagram of a Denil fishway. Reproduced from Thorncraft and Harris (2000).

Alternative approaches: implementing dual fishways

In certain cases, passage effectiveness, and the associated capital costs, can be optimised by implementing two separate fishways with distinct ecological and hydrological functions. An example illustrating this approach has been observed at the Wentworth Weir on the Murray River, Australia (O'Connor et al. 2017b). Here, a VSFW has been installed to accommodate the passage of small-, medium-, and large-bodied fish during low to medium flows. However, during high flow periods, when larger fish species such as Golden Perch and Murray Cod are migrating, the VSFW is not suitable. To avoid the significant expense of extending the operating range of the VSFW to accommodate high flows, a second fishway was constructed. This alternative fishway takes the form of a short Denil fishway, which offers higher discharge capacity and greater attractiveness to larger fish, thus providing enhanced functionality for their passage during high flow conditions (O'Connor et al. (2017b).

Figure 7-24: A dual Vertical Slot and Denil fishway design on Pipeclay Creek, Australia. Photo credit: Tim Marsden.

Nature-like fishways

Nature-like fishways have a range of applications and are suitable for barriers with relatively low hydraulic head. These fishways mimic natural stream characteristics in a channel that bypasses the dam. Consequently, these designs are generally suitable for a wide range of fish species and life stages (NOAA 2012). Recent international studies found that even crayfish, molluscs, and other aquatic invertebrates have been found to utilise nature-like fishways (Nielsen and Szabo-Meszaros 2022). Two main sub-types of nature-like fishways are summarised here based on the predominate construction material and style of the flow control structure between the pools:

- Rock ramp fishways
- Bypass channels

Rock ramp fishways

Rock ramp fishways are generally constructed inside the stream channel, butting against and over the pre-existing dam. They may be full or partial width. Rock ramps can be suitable for structures where the hydraulic head is up to 4 m, but generally their applicability to structures > 4 m height and/or with > 4 m hydraulic head is limited and will be context/site-specific. A key limitation of rock ramp fishways on larger structures is that they normally do not allow for low headwater levels. Like nature-like bypass channels, rock ramps have variable passage efficiencies (Franklin et al. 2012; Stoller et al. 2016) and there is little evaluation of their performance at low-head dams globally (Haro et al. 2008). They have, however, proven effective at enabling multi-species passage, including for small-bodied species and life stages, at low-head (<4 m) structures in Australia and are considered a good option for providing passage at lower head structures in New Zealand. For further details on design criteria for rock ramp fishways see Section 5.5.3.

Nature-like bypass channels

Nature-like fishways have a range of applications and are suitable for all barriers, if there is sufficient space to construct the fishway while maintaining an appropriate gradient and shape (Figure 7-25). Nature-like bypass channels are particularly useful for upgrading existing installations. This type of fishway is generally considerably cheaper to construct than technical fishways. They are negotiable by most fish species and blend into the surrounding landscape.

Nature-like fishways are often designed as bypass channels to reroute part of the water around structures and/or circumvent the structure completely to restore longitudinal connectivity (Tamario et al. 2018). Nature-like fishways are generally well suited for installation at small dams (<3 m height) because larger dams can require a prohibitively large fishway footprint due to the low gradient of the fishway (Katopodis et al. 2001). In the New Zealand context, nature-like by-pass channels might be more appropriate for dams with a height lower than 4 m, but where sufficient space is available are also a viable option at higher head structures.

In general, the channel needs to be well armoured and as diverse as possible and should include natural characteristics such as resting pools, riffles, runs and backwaters (Thorncraft and Harris 2000). By including channel diversity, a range of velocities will be provided within the channel, but it is essential that these velocities are within the sustained swimming speed of weak-swimming fish, with only a few areas where burst swimming would be required. It is also important to maintain a low channel gradient (e.g., 1:40–1:50) and shape, so that at both low and high flows, low velocity wetted margins remain available for fish passage (O'Connor et al. 2017b). In catchments prone to extreme water fluctuations, the channel should be able to cater for the range of flows that exist. Wherever possible, different sized material (including woody debris) should be used in the construction. Pool and riffle spacing of six times the channel width and a meander of 12 times the channel width have been recommended (Newbury et al. 1998), but should be adjusted relative to and informed by local geomorphology. The banks should be planted to provide shade as well as maximise flood protection and in-stream cover.

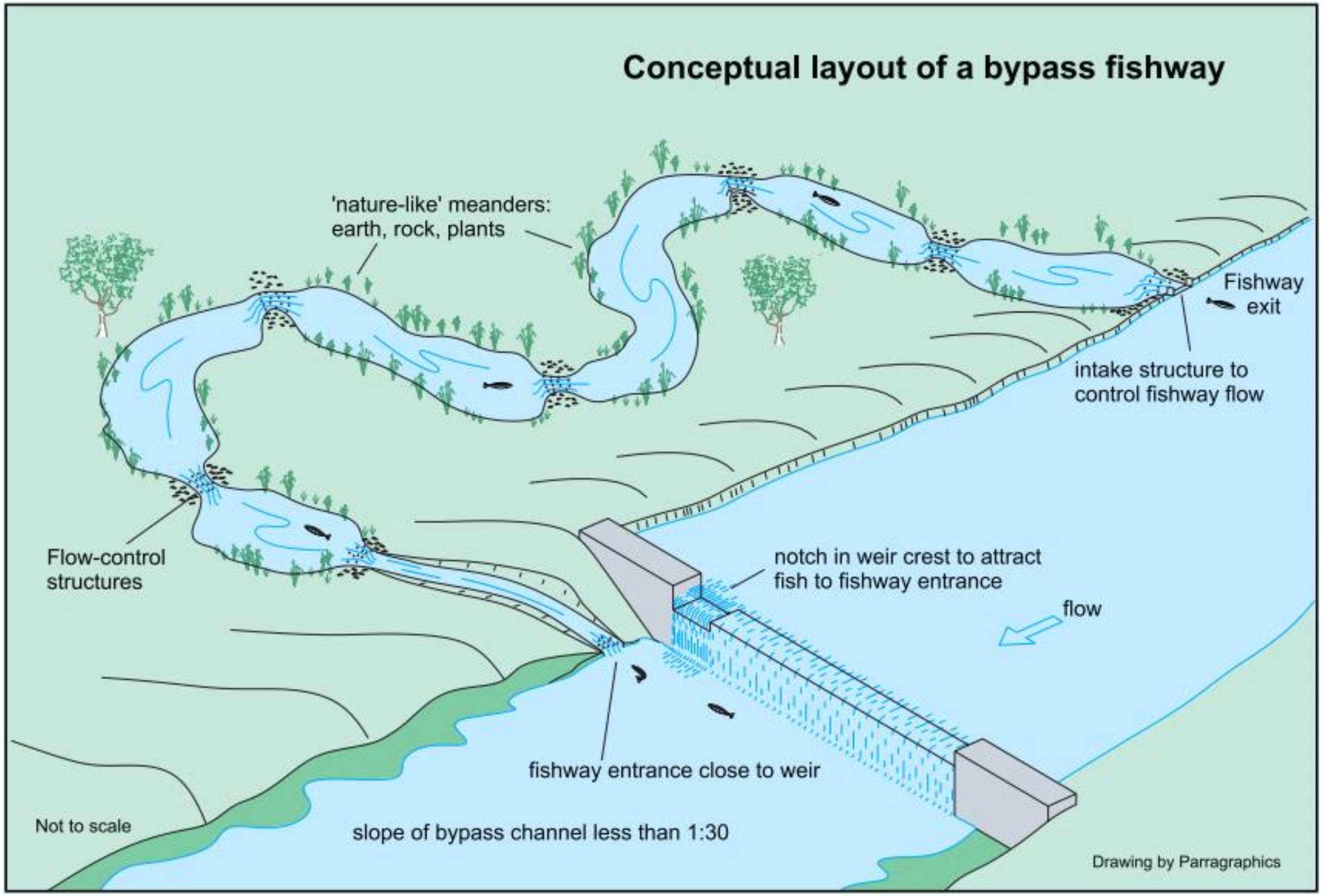


Figure 7-25: Conceptual diagram of a nature-like bypass fishway that could be constructed to allow fish passage past a dam. Reproduced from Thorncraft and Harris (2000).

The technical design recommendations below for rock fishways like nature-like bypass channels target small-bodied fish (25–150 mm long, including species like īnanga), as these are the weakest swimmers. Because each site and fish community are unique, it is suggested that the individual criteria are refined as part of a collaborative process for any new or retrofitted fishways. Further specifications for medium- to large-bodied fish are found in O'Connor et al. (2017b).

Table 7-3: Recommended physical and hydraulic standards for small-bodied fish (e.g., īnanga) for rock fishways including nature-like bypass channels (up to 6 m head differential; O'Connor et al. (2017b).

Specifications	Design guidelines
Fishway operating range and differential head	The range of flows and differential head over which the fishway operates is a site-specific decision, but the standard criterion of fishway operation up to and including a 1-in-5-year flood is a baseline requirement.
Resting pool	Resting pools are typically used for every 1 m rise in vertical elevation and some designs include oversized resting pools with a range of habitats that provide habitat complexity, and associated protection from predation and resting habitat for all expected species and size classes.
Pool size	The recommended generic pool size for a ridge-style rock fishway is 2 m long (clear space), allowing dissipation of flow to maintain acceptable turbulence levels and appropriately quiet water in fish resting areas. Pool size may be reduced where head loss is also reduced.
Minimum depth	The minimum depth recommended for small-bodied fish is 0.3–0.4 m in at least 50% of the pool area in a continuous path.
Slope	A slope of 1:30–1:50 (vertical: horizontal) is recommended for the passage of small-bodied species, but there is scope to steepen the fishway where head loss and turbulence are low.
Head differential	The head differential for a bypass channel is a site-specific decision, but 75–100 mm (i.e., corresponding to velocities of 1.0–1.22 m s^{-1}) is a starting point for many rock fishways, depending on the fish species present. We suggest that no head loss should exceed 120 mm.
Hydraulics	Bypass channels must provide 'hydraulic diversity' so that fish can choose their ascent path. Turbulence should be minimised, with little 'white' water in the fishway pools, and if there is an assumption that turbulence can be calculated in the same manner as for a vertical slot, then it should be 25 W m^{-3}.

The performance of nature-like fishways as a bypass solution and/or integrated with other solutions has been evaluated largely for salmonids (Nyqvist et al. 2018; Raabe et al. 2019) and other diadromous species (Landsman et al. 2018), but to date there has been limited focus on their effectiveness for species such as eels and/or lamprey. Poor attraction is a common problem associated with nature-like fishways, although passage efficiency can be high (Bunt 2001). Low passage efficiencies can be attributed to inadequate flows or poor attraction efficiency (often due to poor siting of the fishway entrance).

As with all solutions, it is important to ensure that a nature-like fishway is functioning correctly, and to initiate a regular monitoring programme to ensure performance measures are met. This could include visually inspecting the channel to ensure that the original channel design has not been moved during floods and undertaking ecological monitoring and associated hydraulic measurements. A benefit of nature-like fishways is that they also provide habitat for fish and can often support resident fish populations.

7.3.6 Novel Solutions

While there has been success in providing fish passage at smaller low-medium head dams through conventional solutions, fishway performance, particularly for large dams, is severely deficient globally (Bunt et al. 2012; Noonan et al. 2012; Hershey 2021). The current fish passage options available for large dams include mechanical designs such as fish lifts and locks (see Section 7.3.4). However, these designs often suffer high costs and poor efficiency and reliability due to operational issues. Trap-and-transfer systems have shown effectiveness at some dams (e.g., Karāpiro), but they can be limited due to their reliance on manual operations by on-site personnel and little effort has to date been put towards optimising the efficiency of trap-and-transfer schemes in NZ. Overall, high costs and poor performance are key factors at preventing the installation of fish ways at large dams (Thorncraft and Harris 2000; Nielsen and Szabo-Meszaros 2022). To address the effectiveness and high costs of currently used conventional methods for fish passage, renewed efforts are being made globally to develop alternative solutions for multi-fish transport across high-head structures. However, many of these systems remain largely experimental with limited evidence for their long-term effectiveness. They are included here for completeness rather than as a recommendation for use.

Whooshh Fish Transport System

An emerging technology in North America is the Whooshh Fish Transport System (WFTS), developed by Whooshh Innovations, LLC Seattle, Washington, which uses differential air pressure to propel fish inside a low-friction, flexible tube over heights >80 m (Geist et al. 2016; Garavelli et al. 2019). The WFTS has demonstrated successful, volitional entry and autonomous passage for Pacific salmonids along the west coast of the United States (Mesa et al. 2013; Geist et al. 2016; Garavelli et al. 2019). In a pilot study, it has also shown successful passage of teleost species in the Great Lakes (Zielinski and Freiburger 2021). The advantages of the WFTS include scalability and the ability to operate irrespective of hydraulic conditions. Currently efforts are underway to pair the WFTS with an imaging hood that captures multiple photographs, allowing automatic identification and sorting of fish before entering the WFTS (Garavelli et al. 2019).

Although this technology has proven effective at some large dams in the Northern Hemisphere, it has not yet been trialled in the Southern Hemisphere due to being primarily designed for transporting large adult fish. In addition, the attraction method used by the WFTS is not well-suited for New Zealand native fish species, which lack the swimming abilities of their North American counterparts.

Tube Fishways

Like WFTS, tube fishways are an evolving closed-conduit fish passage solution (Peirson et al. 2021) designed for Australian native fish species. The design consists of a self-powered water propulsion system suitable for lifting fish over high structures (Peirson et al. (2021); Figure 7-26).

The operation of the tube fishway involves two main phases: (a) volitional attraction of fish into the transfer chamber located in the downstream reservoir of a dam and (b) non-volitional lifting of fish with an unsteady surge at near atmospheric pressures up and over dams (Peirson et al. 2021).

Trials in Australia have been successful for structures ≤8 m, and the design has shown promising laboratory results in terms of fish attraction (Harris et al. 2019; Farzadkhoo et al. 2022), injury-free lifting of fish (Peirson et al. 2022) and automated continuous operations (Felder et al. 2022). These results also indicate that the tube fishway could potentially be scaled up to dams over 100 m high using its innovative lifting mechanism (Peirson et al. 2021). Although tube fishways have not yet been installed, prototypes exist, and passage of Australian bass (~ 50 mm) and rainbow trout (~180 mm) have been tested with positive results (Harris et al. 2019).

A significant benefit of the tube fishway is its ability to enable upstream passage for fish of a variety of sizes, regardless of swimming ability. However, there is currently no data available to support the suitability and effectiveness of tube fishways for New Zealand species. Further research is required to assess the applicability of this fish passage solution for New Zealand.

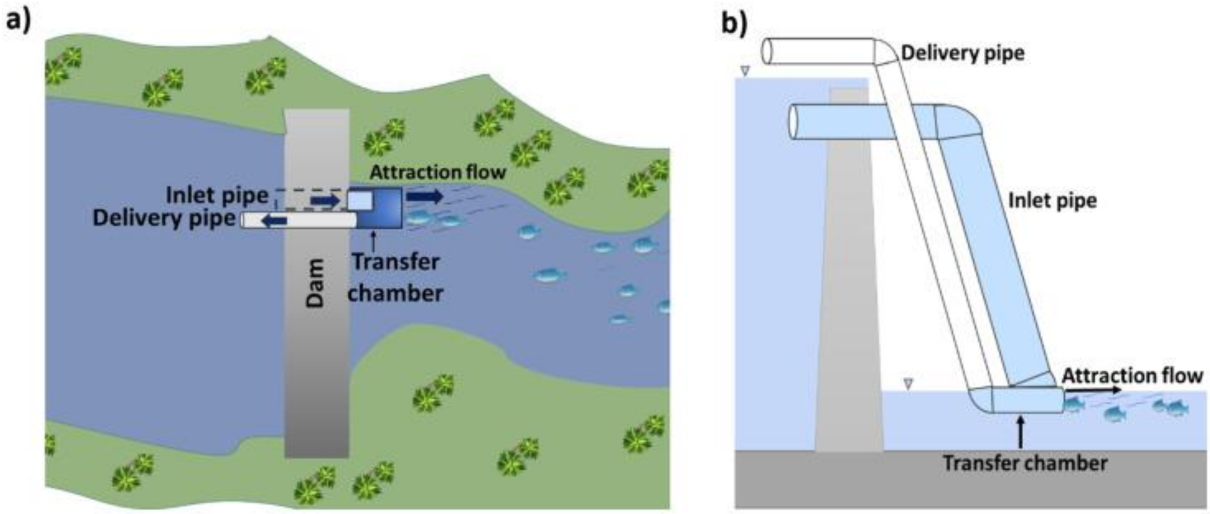


Figure 7-26: Diagram of the Tube Fishway across a dam: top view (a) and side view (b). From Farzadkhoo et al. (2022).

Archimedes Screw Lift

Studies have shown that Archimedes screws, known for their fish-friendly attributes, may offer a viable alternative for facilitating safe upstream fish migration (Zielinski et al. 2022). The device uses a rotating helical blade to extract energy from the water (i.e., turbine) or lift water continuously (i.e., pump, Koetsier and Blauwendraat (2004), thereby continuously lifting fish and water over low-head barriers. Internationally, high survival has been demonstrated during upstream passage trials for multiple fish species. Notably, McNabb et al. (2003) found survival rates of 93–98% in juvenile Chinook salmon (*Oncorhynchus tshawytscha*) navigating upstream passage through a 3 m diameter by a 11.5 m long Archimedes Screw Lift. In another study, Vriese (2009) reported no mortalities among nine European freshwater species (N = 99) upstream in a pilot test of a 0.7 m diameter screw. Moreover, Zielinski et al. (2022) built a field-scale Archimedes Screw Lift prototype at the Cheboygan Dam, Michigan, which safely transported 704 fish (688 of which were Catostomidae) in 11 days.

There were no observed injuries in the transported fish or mortalities in a subset of fish held post-transport. The findings of low mortality support the further advancement of Archimedes screws to facilitate upstream passage. However, the application of Archimedes screws as a fish passage solution has not been widely investigated and requires further research to support the suitability and efficacy for New Zealand species of a range sizes and swimming abilities. See section 4.6.3 Pump Design for more details on Archimedes Screw designs.

7.4 Solutions for downstream passage

In contrast to the extensive understanding and implementation of upstream migration solutions, the provision of safe downstream fish passage at low- and high-head dams remains a major challenge, having received less research and development internationally and in New Zealand. At hydropower dams, migratory and non-migratory fish moving downstream can be injured or killed by impingement on intake screens, turbine blade strikes, barotrauma, or physical damage and/or mortality from travel over spillways. At dams that are not used for hydropower, impacts on fish populations include, the attenuation or loss of triggering factors leading to an absence of or delay in migration, and extra energetic costs of the additional distance travelled as result of exploring the reservoirs to find other escape pathways.

Downstream mitigation for fish passage at dams focuses on three primary objectives:

- transporting fish safely downstream,
- preventing fish from being drawn into turbine intakes at hydropower facilities, and
- facilitating the safe and timely movement of fish through reservoirs.

When designing facilities to assist the downstream passage of fish, clear objectives and defined performance measures should first be established (Section 3). With clearly defined objectives, there are several key factors that need to be considered to ensure downstream passage is effective, including fish swimming ability, and behaviour, as well as hydraulic conditions, such as water depth, velocity, and turbulence. Additionally, the strategic placement of entrance locations is critical to downstream migration, as it influences the ease with which fish can locate and access fish passage facilities.

To help downstream passage of fish species, a range of solutions have been developed internationally, with some implemented in New Zealand. Common mitigation strategies include:

1. Preventing turbine entrainment for downstream migrating fish.

A. Trap-and-transfer operations.

B. Installation of fish-friendly turbines.

C. Physical barriers, such as trash racks.

2. Allowing fish to pass through or over a dam.

A. Bypass systems.

B. Operational changes at hydropower dams, such as turbine shutdown and fish passage via spillways.

C. Screens that exclude or ‘guide’ fish (including behavioural deterrents) to a sluiceway or bypass system around the dam and away from turbine intakes by means of manipulating hydraulic conditions.

In New Zealand, the largest concerns around downstream fish migration relate to anguillid species – particularly longfin eel (Case Study 9). Longfin eel penetrate further inland than other large species and many of New Zealand’s hydro-electric dams are located inland to take advantage of more pronounced gradient changes in the landscape when building dams. Like longfin eel, lamprey penetrate well inland and are also a species being transferred upstream of hydro-electric dams as part of trap-and-transfer operations. Should lamprey successfully breed and create resident larval populations upstream of dams, then downstream passage of juveniles (macrophthalmia) would be required. Presently, no larval lamprey have been recorded above any high head dam across New Zealand. As such, the following section focuses on anguillid eels, which have well established populations above high head dams nationwide and safe passage downstream is critical to completing their lifecycle.

Case Study 9: Iwi/hapū-driven downstream tuna passage activities at hydroelectric power stations

Tuna mortalities due to HEPS are a long-felt source of frustration, anger, devastation, and trauma for iwi/hapū communities. For example, during the Te Ika Whenua hearings, Cletus Maanu Paul expressed “tuna cannot descend downstream to breed in Te Moana-nui-a-Kiwa because the dams mince them up in the turbines.” The Te Ika Whenua report outlines the intensity of claimants’ concerns over “one of their precious gifts being forcibly taken from them” and how “no effort was being made to create a diversion for the eels to migrate down to the ocean to breed” (Waitangi Tribunal 1993). It goes on to state that “the effect on the rivers and the claimants’ food sources, particularly eels, are said to be, in their terms, disastrous” (Waitangi Tribunal 1993).

Immediately addressing the significant loss of tuna heke/migrant eels as they pass through HEPS and flood control schemes throughout Aotearoa New Zealand is of utmost importance to iwi/hapū (LMK Consulting Ltd 2014; Williams et al. 2017). As Ken Mair, then chair of Te Wai Māori Trust, said in an address to the 2017 Māori Tuna Conference in Whanganui “We must find solutions to improve waterways to ensure changes are not at the expense of the status and the whakapapa of tuna. It is not a kaupapa we can afford to put off” (Mair 2017).

Iwi/hapū have played a key role in advocating for and initiating efforts to reduce the impacts of HEPS on downstream tuna passage. For example, Ngāti Hikairo ki Tongariro have a strong connection to tuna, which has been impacted by the construction and continued operation of the Tongariro Power Scheme (TPS). John and Lena Morgan (who established Nga Puna Toi Ora Ki Tūwharetoa), reached out to Genesis Energy who provided funding to the whānau and hapū to restore mobility for tuna (Department of Conservation 2019). A trap-and-transfer project was implemented on Lake Otamangakau and Whanganui awa to help the eels get above and below the dams and tunnels associated with the TPS. Elvers are caught in an elver trap and released above the dams for them to find a new home in the upper reaches of the river. Subsequently, large eels migrating downstream are caught and placed below the dam (Wairehu drum screen) to continue their migration to the sea. Similarly, Te Waiau Mahika Kai Trust provided input to the design and implementation of a tuna trap-and-transfer programme for the Waiau catchment run by Meridian Energy (Te Waiau Mahika Kai Trust 2021). Elvers are transferred upstream of the Manapōuri Lake

Control Structure and adult migrants are captured in Lake Manapōuri before being transferred downstream of the scheme.

Unfortunately, as with upstream trap-and-transfer programmes, there has been little effort to monitor the effectiveness of programmes to mitigate impacts on downstream fish migrations. There remains an urgent need for station-specific research undertaken collaboratively by iwi/hapū, fisheries biologists, and HEPS engineers to inform the improved design and implementation of suitable interventions at HEPS.

7.4.1 Downstream passage solutions overview

The following section details key downstream passage solutions that currently exist internationally, including those that are used in New Zealand (for an overview see Table 7-4).

Table 7-4: Overview of the key downstream fish passage facilities available internationally and in New Zealand and their general application. Adapted from Nielsen and Szabo-Meszaros (2022).

Solutions for downstream passage	General applications	Advantages	Disadvantages
Trap-and-transfer (Section 7.4.2)	▪ Wide range of catchable species	▪ Potential to transport fish across any size or multiple dams ▪ Low capital cost, low maintenance, no constrain by power station head ▪ Ability to closely monitor fish ▪ Can manage variable tailwater heights and can operate independently of headwater levels ▪ Relatively small footprint ▪ Can operate with minimal flow (e.g., 1 l/s) compared to technical fishways or bypass channels	▪ Generally, caters only for upstream passage ▪ High operating and labour costs ▪ Likely very poor efficiency in capturing migratory fish ▪ Requires transport infrastructure between the dam's base and upstream impoundment(s) ▪ Often poor understanding of efficiency for different species
Operational Changes (Section 7.4.3)			
Turbine shutdown	▪ All fish species	▪ No need for fish screening systems in front of intakes or a fish bypass system	▪ May terminate power generation during the key migration period ▪ No partial load operation and full turbine shutdown cause revenue losses

Solutions for downstream passage	General applications	Advantages	Disadvantages
Fish passage via spillway and overflow	▪ More suited to robust species and life stages ▪ Better suited to low-head dams	▪ Minimal capital outlay, potentially high attraction flow	▪ Suitability depends upon species stress tolerance, dam design (e.g., dam height, spillway design, tailwater depth), and the potential for revenue loss ▪ Hydraulics and the design of many existing spillways and weirs may not favour fish passage ▪ Weir or Spillway and attraction flow to navigation locks cause hydropower generation and revenue loss
'Fish-friendly' turbines (Section 7.4.4)	▪ Better performance for anguillid fish species, as well as small-bodied and juvenile fish	▪ All flows can pass through the powerplant ▪ Potentially, less need for fish screening systems in front of intakes or a fish bypass system	▪ More data required to support applicability for New Zealand species ▪ Some may cause higher mortality rates in larger fish ▪ Higher cost for ecologically improved turbines ▪ Limited to low-head dams (up to 15–30 m)
Bypass system (Section 7.4.5)	▪ All fish species	▪ Can be tailored to specific fish behaviours and preferences, ensuring that the bypass is effective for a wide range of fish populations ▪ Can be integrated with other fish passage structures, such as guiding devices (trash racks) and fish locks	▪ Can be costly to design, construct, and maintain ▪ Bypass efficiency depends on the available discharge, bypass dimensioning, spatial proximity to fish also guiding structures, location in the water column and prevailing hydraulic conditions ▪ Revenue loss from water bypassing turbines

Solutions for downstream passage	General applications	Advantages	Disadvantages
Trash racks (Section 7.4.6)	▪ All fish species	▪ Can be relatively simple and cost-effective compared to other passage solutions ▪ Depending on the type, they offer high protection and guidance efficiencies for small to large fish species	▪ May not provide total fish protection, as fish smaller than the bar opening can pass through the rack ▪ In retrofitted cases, bypassed flows cause hydropower generation and revenue loss ▪ Depending on spacing, not suitable for large dams due to velocity limitations for fish injuries and potential operational issues due to debris clogging
Screening (Section 7.4.7)	▪ All fish species	▪ Potentially a complete barrier to prevent passage through the turbine ▪ Well-designed bypass systems provide high migration efficiency	▪ Screens blocked by debris increase head-loss ▪ During retrofitting facilities, bypassed flows cause hydropower generation and revenue loss ▪ High costs for screens and potential operational issues due to debris clogging at large hydropower plants

7.4.2 Trap-and-transfer

Trap-and-transfer is currently the most widely adopted downstream fish passage solution at high-head structures (especially hydro-electric schemes) in New Zealand. Trap-and-transfer is regarded as a species-specific solution and globally occurs almost exclusively for eel and salmonid species (Schwevers and Adam 2020; Kock et al. 2021). This solution relies on the effective capture of as many of the target species and/or life stages as possible and their transportation below barriers.

Commercial and/or customised fishing nets are typically deployed to capture target species and this capture method is particularly suitable for still waterbodies. Logistical and practical constraints mean trap-and-transfer can be impractical in large rivers and at intake structures. High capture effort is typically required to implement effective downstream trap-and-transfer and a thorough understanding of the target species' abundance, distribution, population structure, onset and duration of migrations and behaviours in the source waterbody is required to optimise this solution (Stuart et al. 2019; Piper et al. 2020). Furthermore, identification of areas where fish congregate prior to migration is essential to optimise capture efficiency (Jellyman and Unwin 2017). However, many downstream trap-and-transfer activities in New Zealand are not informed by site-specific studies and some hydro-electric schemes do not routinely capture downstream migrant fish during their migration season (Williams et al. 2022). Differences in the onset and duration of downstream migration of different species exist throughout New Zealand (including eels; Stuart et al. (2019))

meaning it is challenging to generate a national guideline for start and end dates for downstream trap-and-transfer activities.

Downstream trap-and-transfer is generally considered a viable temporary passage solution until appropriate and effective downstream passage solutions are implemented (i.e., engineered fish passes etc.; Schwevers and Adam (2020). Unlike upstream trap-and-transfer solutions, there is limited understanding of the national scale of downstream trap-and-transfer activities in New Zealand because there is no nationally co-ordinated programme, and access to data is limited (Williams et al. 2022). Evaluating the effectiveness of downstream trap-and-transfer is particularly challenging and requires intimate knowledge of the site-specific ecology and behaviours of the target species (Williams et al. 2022). Furthermore, the effectiveness of trap-and-transfer activities can vary widely between years due to variable environmental conditions over various timescales that ultimately influences fish catchability and total numbers/biomass transferred downstream (Stuart et al. 2019; Bourgeaux et al. 2022).

The unique operating regimes of hydro-schemes can influence catch-rates and knowledge of how each operating regime (e.g., spilling frequency, lake level fluctuations) influences migrant trap-and-transfer activities is imperative to success (McCarthy et al. 2008). Quantifying success of trap-and-transfer as a downstream passage solution is generally more achievable when completed in rivers compared to lakes (Piper et al. 2020).

Dams that are not used for hydropower generation (usually with a spillway) have previously been considered 'safe' for adult migrant eel passage, but negative effects are increasingly documented (Trancart et al. 2020). Five main impacts on eel populations are (i) the attenuation or loss of triggering factors leading to an absence of, or delay in, migration; (ii) extra delays and extra distances travelled when crossing the dam; (iii) extra energetic costs of the additional distance travelled as result of exploring the reservoirs to find other escape passages; (iv) the selection of a more risky behavioural phenotype, i.e., bold eels; and (v) direct blocking of migration pathways once migration has started (Trancart et al. 2020; Bourgeaux et al. 2022).

Guidelines for downstream trap-and-transfer programmes include:

- Clearly identifying objectives and measures of success (Section 3).
- An understanding of migrant shortfin and longfin eel behaviours so that the siting of trap-and-transfer activities optimises capture (Jellyman and Unwin 2017).
- Implementing a robust monitoring programme to assess the effectiveness of trap-and-transfer operations. This can involve tracking fish movements, survival rates, and population dynamics both upstream and downstream of the barrier.
- Considering the timing and frequency of trap-and-transfer operations in relation to fish migration patterns. Coordinate the operations to coincide with peak migration periods to maximise the number of fish captured and transferred.
- Selecting suitable transportation methods for transferring fish downstream. This may include using specialised fish transport trucks, barges, or bypass channels designed to provide safe and efficient passage for fish.
- Identifying appropriate release locations downstream of the barrier that provide favourable conditions for fish survival and continued migration. Consider factors such

as water temperature, flow rates, and the presence of suitable habitat for the target species.

7.4.3 Operational changes (or 'Fish Friendly' Operational Management)

Implementation of downstream passage at hydro-electric dams can be complex and expensive (Larinier and Travade 2002). Consequently, alternative solutions, such as turbine modulation/shutdown during migration peaks, and spillway availability are increasingly being considered to mitigate the impact of dams on fish communities (Thorstad et al. 2012; Stich et al. 2015; Teichert et al. 2020). Operational changes can be adopted to complement and/or enhance existing downstream passage solutions such as trap-and-transfer and are rarely applied as a stand-alone solution. It is also important to note that, for these methods to be effective, it is necessary to have an appropriate spill bypass design, as fish are more likely to be attracted to this bypass if it is close to the power intakes (Nielsen and Szabo-Meszaros 2022).

Active spillway releases

Active spillway releases (as opposed to natural events during very high rainfall) can be a viable option to enhance downstream passage success during large migration events occurring at off-peak generation times (e.g., silver eels migrating at night). By considering trade-offs between the cost of lost generation and the implementation of physical or behavioural deterrents, site-specific measures can be employed. It is crucial, however, to ensure the spillway design enables safe passage of migrant fishes. Overtopped or partially opened spillways may present alternative routes to turbine passage under high flow conditions when the proportion of flow being abstracted by turbines is minimal compared to total discharge (Adams et al. 2014; Fjeldstad et al. 2018).

A field trial conducted by Watene and Boubée (2005) at Patea Dam (82 m high) revealed that more than 70% of tagged migrant shortfin eels (850 to 940 mm) introduced under the opened spillway gates with a 70 mm opening and a head of around 9 m, survived with minimal injuries. It is worth noting that some of the injuries observed on the released fish were likely a result of capture, storage, and handling, indicating that the actual survival rate would typically be higher than the 70% suggested by the trial. Although eel survival rates can vary depending on the site, this study highlights the viability of spillway releases as an effective measure for improving downstream passage success for migrant eels. However, it is important to acknowledge that while fish passage over spillways or through outlet discharges is considered a safer alternative to turbine passage, there is still some injury and mortality occurring (Coutant and Whitney 2000).

Turbine shutdowns

Downstream fish passage through turbines can be further reduced by total or partial turbine shutdowns. This downstream solution can in principle be effective but can become very costly for the hydroelectric operator if the shutdowns are not targeted. The challenge is not only to target, but also to anticipate the events of downstream migration (Dewitte 2018).

Early warning systems can be implemented to facilitate the detection and anticipation of downstream fish migrations, enabling proactive measures to be taken in a timely manner. These systems utilise various sensors, such as DIDSON sonars, MIGROMAT, and water quality sensors, to identify indicators of mass fish migration events. Once detected, the system alerts the hydropower plant operator, allowing them to switch to a fish-friendly mode of operation. In certain instances, this may necessitate the temporary suspension of power production during critical migratory periods (Schwevers and Adam 2020).

Manawa Energy upgraded the downstream migrant eel bypass facility at Patea Dam in 2015, which utilises a purpose built bypass and operational management through turbine shutdowns. Between 1 March to 31 May each year, when rainfall triggers are met, generation is shut down overnight and either the spillway is opened or the bypass facility is used to maintain the minimum flow over Patea Dam and provide safe passage downstream for migratory eels (Goldsmith 2018; Fern 2019). The bypass facility diverts water from the intake for the auxiliary unit to a diverter box where migrant eels can be held for counting and releasing the following day (Goldsmith 2018). As generation has ceased, the only flow over the dam is either over the spillway or through the bypass intake, which is located at the entrance to the penstocks. In 2016, 311 migrant eels used the bypass with 54 passing the dam over the spillway (Goldsmith 2018).

Models incorporating environmental parameters such as temperature, turbidity, and velocity have also proven effective in identifying migration events of specific species. For instance, a turbine shutdown model was developed and refined over a six-year period at a hydropower station located on the Dordogne River in France (De Oliveira et al. 2015). During downstream eel migration periods, turbine shutdowns were implemented for 50 to 77 nights throughout the study (Labedan and Sagnes 2021). The effectiveness of the model varied between 70% and 100% depending on the year and the indicator used, whether it be catches reported by the fishery or radio-tracked eel passages. Another notable example involves the use of predictive models derived from telemetry studies on eel migration to establish decision rules for turbine management in the Shenandoah River system (Smith et al. 2017). By establishing relationships between migrant eel catches and environmental data, certain hydro-companies are now able to minimise mortality rates of migrating eels, optimise capture rates, and minimise impacts on power generation by following specific decision rules. For instance, one decision rule implemented is temporarily halting turbines when lake levels increase by 10 cm to minimise eel mortalities (Teichert et al. 2020).

7.4.4 Turbine replacement

Hydro-electric power stations employ various types of turbines (e.g., Pelton, Francis, Kaplan, Bánki-Michell Bulb and Deriaz) designed to exploit a range of different flows and head heights. These turbines can pose risks to fish, potentially leading to severe injuries and mortality (Algera et al. 2020). The extent of the impact on fish, in terms of injury and mortality, depends on several factors, including specific characteristics of turbine type (head-height, turbine rotation speed, runner blade number, blade gap), flow, and fish (species, size, behaviour, and physiological conditions; e.g., Mitchell and Boubée (1992), Larinier (2008), Radinger et al. (2022)).

In New Zealand, most hydro-electricity is generated using Francis turbines, while some power stations, like the Karāpiro dam, employ Kaplan turbines, and others like the Waipori power scheme employ Pelton turbines. Francis turbines typically have more blades and are common at high-head sites (Larinier 2000; Figure 7-27). Kaplan turbines have a propeller type design with variable pitch blades that allow the turbines to be operated across a range of flows (Trumbo et al. (2014); Figure 7-27). Although New Zealand-specific studies on turbine mortality or blade-strike models are lacking, international research suggests that Kaplan, Francis, and Pelton turbines generally exhibit among the highest rates of fish mortality and injury compared to other turbine types (Larinier and Travade 2002; Wilkes et al. 2018b; Algera et al. 2020). Of these, Kaplan turbines are associated with the lowest mortality rates across all fish species (Pracheil et al. 2016). Due to their elongate body shape, anguillid species, like New Zealand's longfin and shortfin eel, are particularly vulnerable to risks associated with turbine passage (Mitchell and Boubée 1992; Beentjes et al. 2005; Watene and Boubée 2005).

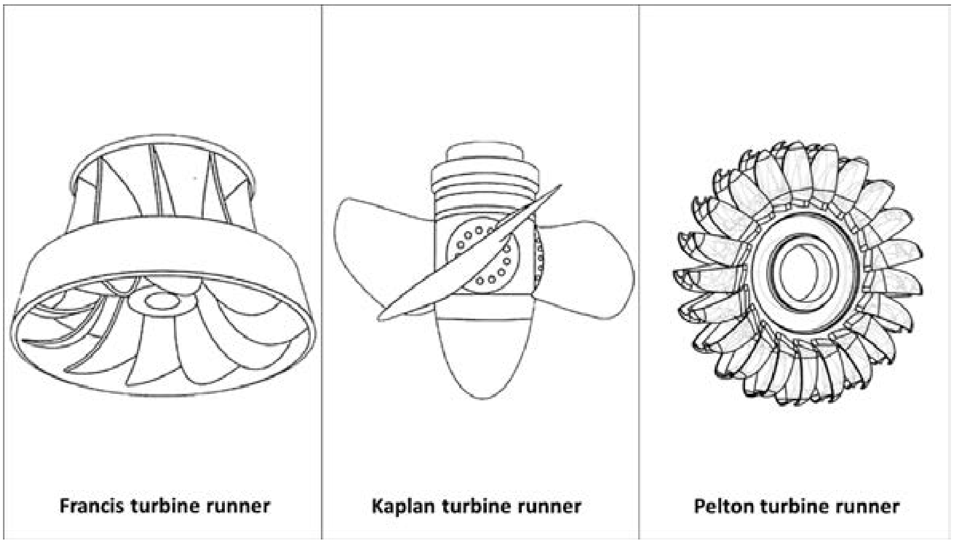


Figure 7-27: Schematic of a conventional Francis, Kaplan and Pelton turbine runners used in New Zealand.

'Fish friendly' turbines

To mitigate risks and improve downstream fish passage, the development of 'fish-friendly' turbine technologies is considered the optimal solution. Internationally, substantial efforts have been underway to develop and test improved turbine designs with enhanced geometry and operating features that aim to minimise potential sources of injury to fish. It is important to note that the term 'fish friendly' is relative, as achieving zero mortality with turbine replacement alone is challenging (Mueller et al. 2022). As such, while 'fish-friendly' turbines are a valuable measure, they should be considered in combination with effective up and downstream fish migration facilities when addressing passage at hydro-electric dams. In the United States, the US Environment Protection Agency has established specific considerations and engineering-based criteria that are considered critical in the design of 'fish-friendly' turbines; these criteria involve (Cooke et al. 2011; Algera et al. 2020):

- Blunt, thick blades (see specifically the Restoration Hydro Turbine (RHT));
- Low rotational speed of the turbine runner;
- Large passages;
- Few blades;
- No exposed gaps.

The following includes a summary of turbine types that have been classified as 'fish friendly' turbines at dams by international experts (Nielsen and Szabo-Meszaros 2022). These turbine types are currently available, or in various stages of development and testing internationally (Figure 7-28; Table 7-5).

The **Alden turbine** is an improved ecological version of the Francis turbine, suitable for medium head applications between 10 and 40 m head, and for flow rates above 25 $m^3 s^{-1}$ (Figure 7-28; Table 7-5). The turbine features a lower rotational speed, reduced number of blades compared to conventional turbines and an altered geometry of the blades for thicker leading edges wrapped around the vertically rotating shaft. The design reduces shear forces, pressure fluctuations and cavitation and the absence of gaps minimises fish entrapment risks (Hogan et al. 2014). Computational Fluid Dynamic simulations and experimental tests at a pilot scale in the Alden research laboratory demonstrated high hydraulic efficiency and fish passage survival rates exceeding 98% for eels up to 430 mm long (Hogan et al. 2014; Dixon and Hogan 2015). It is important to note that some New Zealand eels migrate at a significantly larger size than those tested so far. Furthermore, this technology has yet to be demonstrated in a field application.

The **Minimum Gap Runner** (MGR) is a recent development of the Kaplan turbine (Figure 7-28; Table 7-5). The design reduces clearances between the adjustable runner blades, the hub, and the discharge ring, minimising risks posed by the flow passages of conventional turbines. MGR turbines are typically used in hydraulic heads ranging of 10 to 25 m and flow rates exceeding 17 $m^3 s^{-1}$ (Dewitte 2018). Despite higher development costs, MGR turbines have shown improved fish survival rates (from 88% to 95%,) and overall efficiency compared to conventional designs (Albayrak et al. 2014). MGR technology has been implemented in major hydro-electric power plants in on the Columbia River, USA (Albayrak et al. 2014).

The **Restoration Hydro Turbine** (RHT) is suitable for installations with a head range of 2 m to 10 m, accommodating retrofit and new systems (Figure 7-28; Table 7-5). RHT's design reduces the risk of fish entrapment through a combination of factors such as low fish length to blade thickness ratio, and reduced blade speeds. The absence of a convergent gap at the blades' connection to the rotor hub further minimises entrapment risks (Amaral et al. 2020). The RHT was developed in 2019, and the first field installations took places in the USA in late 2019 and 2020. Recent field studies have demonstrated the efficacy of RHT, specifically that at 10 m head, 100 % of tested American eels (n = 131), measuring between 33.9 and 65.5 cm, safely passed through the turbine (Watson et al. 2022).

The **Very Low-Head turbine** (VLH) is a specialised type of Kaplan turbine inclined at an angle (typically 45°) designed to optimise power production efficiency for discharges up to 30 $m^3 s^{-1}$ at low-head sites (up to 4.5 m), while minimising impacts on fish (Wright and Rival (2013); Figure 7-28). Fish-friendly design features include a large diameter runner (4.5 m) with wide spaces between blades, low runner speed, reduced pressure variations, and minimised gaps to prevent fish entrapment (Lagarrigue and Frey 2011). Fish passage tests conducted in 2008 using 150 European eels showed a 95% survival rate for adult eels (700 mm to 1200 mm) passing through the VLH turbine. Subsequent tests with a second generation of runners showed no mortality for smolts (232 mm) and eels (760 mm) (Lagarrigue 2013).

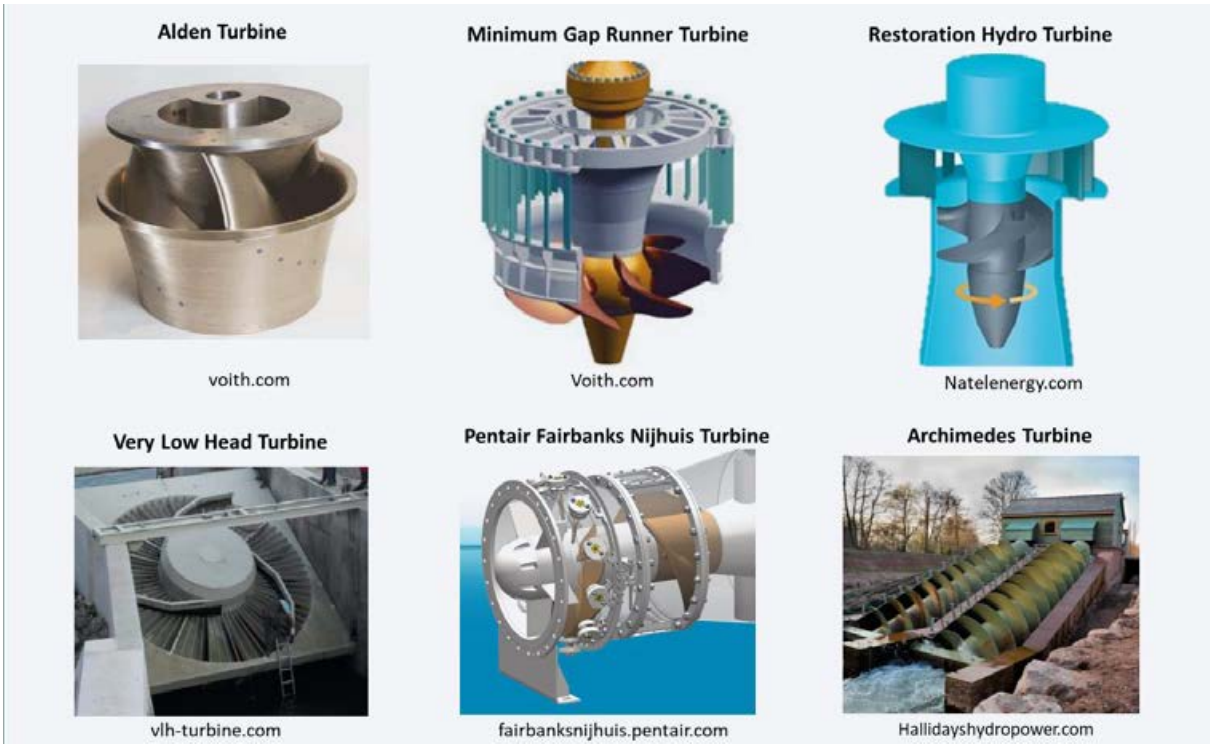


Figure 7-28: Examples of Fish friendly turbines currently used and in development internationally.

Table 7-5: Basic damage potential of turbine types. From Dewitte (2018).

Turbine type	Strike probability	Impact fatality	Pressure load	Shear and turbulence	Overall mortality
Francis	High-medium	High-medium	Low	High-medium	High
Kaplan	Variable	Variable	High-medium	Low-medium	Variable (high for anguillids)
Pelton	High	High	-	High	Lethal
Alden	Low	Low	Low	Low	Low
MGR	Low-medium	Low-medium	Low	Low	Low
VLH	Medium	Low-medium	Low-medium	Low	Low-medium
RHT	Low	Low	Low	Low	Low
Pentair Fairbanks Nijhuis	Low	Low	Variable	Low	Low
Archimedes Turbine	Medium	Medium	Low	Low	Low-medium

Pentair Fairbanks Nijhuis (Low Pressure) turbines, like the Alden turbine, are characterised by their two or three helical blades with rounded front edges (Figure 7-28; Table 7-5). The design has an even pressure profile and prevents cavitation across the turbine's operating range (van Esch and van Berkel 2015).

These turbines can be installed either vertically or horizontally and are suitable for head heights up to 8 m and flow rates of 1.5 to 150 $m^3\ s^{-1}$. Tests conducted by the University of Wageningen used smaller scale turbine models (1:5), with eels measuring an average of 400 mm, showing low mortality rates even 96 hours after turbine passage (Winter et al. 2012). Another study, on a further down-scaled model (1:16; discharge of 0.7–0.8 m^3/s and head of 0.51–0.82 m) at the Nederrijn River, found no mortality or injuries to eels with average lengths of 250 mm (Vriese 2015).

The Archimedes turbine is a simple design, well suited for low head (<10 m) installations (Figure 7-28; Table 7-5). This turbine consists of a screw-shaped runner, placed within a co-axial, tubular shroud. When water enters the top of the shaft, the water weight pushes on the screw blades, causing the shaft to rotate, allowing for the water to fall to the lower level. Such designs are intended for low heads and have diameters between 1.5 and 3.5 m (Lashofer et al. 2012). The fish-friendliness of Archimedes turbines has been a subject of ongoing discussion. Studies have shown that these turbines have the potential to minimise injuries to fish due to their low rotational speeds and reduced shear forces. However, concerns have been raised about potential second order effects, such as altered fish behaviour prior to or after passage. Piper et al. (2018) examined European eel passage past Archimedes screw turbines across two years at the Flatford Mill on the River Stour. Some eels were delayed upon reaching the turbines and exhibited frequent rejection and milling on their approach (Piper et al. 2018). Although Archimedes screw turbines could potentially provide a safe downstream passage route for eels, the delay to seaward migration and the energetic costs associated with migration delays should be considered. See Section 4.6.3 for details on Archimedes pump designs.

7.4.5 Bypass systems

To reduce the adverse ecological impacts of hydropower plants on downstream moving fish, it is important to establish well-designed and alternative migration corridors or bypass systems. Bypasses should be carefully planned, taking into consideration not only the design and location, but also the unique characteristics of the site-specific fish community. However, it is important to note that standardised design criteria for bypasses in New Zealand are currently lacking and are, therefore, not within the scope of these guidelines.

Bypass efficiency is influenced by multiple factors, including not only the available discharge, but also bypass dimensioning, spatial proximity to fish guiding structures, location within the water column (top, middle, bottom), and prevailing hydraulic conditions at the entrance of the structure (Larinier and Travade 2002; Katopodis 2010). Given the considerable variation of these variables among different sites, it is crucial to account for local conditions.

Placing the escape-route entrance too far from the water intake or barrier significantly reduces passage efficiency (Larinier and Travade 2002). Consequently, location of any potential escape-route entrance is a key priority that should precede bypass design. For migratory eels, sub-surface bypasses are commonly recommended. Studies at Wairere Falls Power Station (Mokau River, Waikato) demonstrated that two 100 mm diameter surface bypass holes drilled side by side in the dam wall, positioned 0.6 m below the water surface provided some passage for downstream migrating eels (Boubee and Williams 2006). Two additional 150 mm entrances have been added since at this site and monitoring undertaken in 2008 suggested that 1,044 migrant eels used the bypasses that year (Stevenson and Boubée 2009).

Bypasses can be implemented alongside physical or behavioural barriers or used independently, provided that appropriate flow conditions naturally guide fish towards the bypass. In addition, fish guidance systems can also be paired with sensory stimuli to enhance efficacy. For example, Deleau et al. (2020) combined an inclined rack with vertical bars (spaced at 12 mm) with an acoustic field to examine avoidance by downstream migrant European eel. The majority of eels reached the bypass in the experiments, with only three eels passing through the screen during the control trials without sound and one passing through during each acoustic treatment (Deleau et al. 2020).

Hydraulic conditions can vary with the design of the associated guidance devices, such as trash racks (Szabo-Meszaros et al. 2018). Trash racks with horizontal bars combined with a bypass can be a preferable solution for fish protection at smaller hydro-power plants, while trash racks with vertical bars can be an alternative for larger schemes (Boes et al. 2016). Ultimately, successful fish bypass schemes depend on monitoring fish migration. This is particularly important if a water spill is needed since bypass success is often related to the magnitude and timing of spill water discharge (Fjeldstad et al. 2012).

7.4.6 Guidance devices

Guidance devices guide or divert fish to bypasses and/or alter or take advantage of natural behaviour patterns. The efficacy of different fish guidance devices varies based on the swimming capabilities and behaviour of the target species and the localised hydraulic conditions. Presently, too few guidance devices have been shown to be effective for downstream migrating eels at small and large head hydro-electric dams. Therefore, standardised design criteria for fish guidance devices in New Zealand are currently lacking and are not within the scope of these guidelines. The following section outlines trash (bar) racks, which can be used as **physical barriers** and **guidance devices** but would require testing for efficacy at New Zealand hydro-electric intakes.

Trash racks as physical barriers

Physical barriers such as bar racks prevent fish species from entering the turbines at hydropower facilities. These barriers are sometimes referred to as trash racks due to their dual purpose, but with proper design they can be effective in the safe downstream passage of migrating fish species. Generally, trash racks are physical barriers used to prevent large debris from entering power station/penstock intakes. Trash racks can provide a certain level of protection for migrating fish by serving as a physical barrier, provided the gap between the bars is sufficiently narrow so that fish cannot pass through the bars (Boes et al. 2016). Conventional trash racks are generally vertical and perpendicular to the flow direction. The profile of the bars is usually rectangular and the bar spacing is typically between 10 and 30 mm (Fjeldstad et al. 2018), although the bar clearance can be up to 100 mm (Meister et al. 2022). In general, the smaller the turbine, the smaller the bar clearance needs to be. In major run-of-river power stations, bar clearance of conventional trash racks is typically between 80 and 200 mm (Schwevers and Adam 2020).

Trash racks that combine ecological and technical requirements are known as ‘fish-friendly’ trash racks, and are one solution to mitigate fish mortality at a low operational cost (Szabo-Meszaros et al. 2018). Fish-friendly trash racks typically have a bar clearance of 20 mm or less (Lemkecher et al. 2021). For run-of-river plants with low design heads, fish-friendly trash racks can cause head losses and significant relative energy production losses (Böttcher et al. 2019). However, in schemes with medium to large head heights, the effects of any head losses associated with fish-friendly trash racks may be lower compared to smaller schemes.

Trash racks with various bar profiles (see below) are now available as a solution to lower any head losses that might be associated with fish-friendly trash racks (Lemkecher et al. 2021).

Design criteria

Bar clearance is one of the most important design parameters to be considered in selecting the most appropriate trash rack. Other design criteria that must also be considered when selecting trash racks for fish exclusion purposes include:

- bar alignment;
- bar angle;
- bar shape/profile;
- velocity.

Bar clearance

To protect fish species, trash rack bar clearance must be smaller than the body dimensions of the target species. Studies have shown that trash racks with 20 mm bar spacing affords protection only for large (>66 cm total length) female European eels. Male European eels sexually mature at a smaller size than females and these migratory male eels (>49 cm) are only protected by trash racks with 10 mm bar spacing (see references in Schwevers and Adam (2020)). Trash rack bar clearance of 12.5 mm or less has been recommended for the European eel (Solomon and Beach 2004), which is smaller in body size than the shortfin and longfin eel. Based on the head widths of New Zealand shortfin and longfin eels, it is anticipated that a 20 mm bar spacing will protect migrant longfin eels and larger female shortfin eels. However, 10 mm bar spacings would be necessary to protect male shortfin eels. The use of trash racks with narrow bar spacing may, however, raise concerns about increased clogging effects and trash rake design.

Bar alignment and angle

The bars on trash racks can be either vertically or horizontally orientated and this can affect their performance to exclude and/or divert fish from infrastructure (as well as the operational efficiencies for hydro-electric power stations). Raynal et al. (2013b) investigated the effect of bar-alignment with vertically oriented ‘streamwise’ bars (i.e., parallel to flow) and vertically angled bars (also called ‘classical’ trash racks) on head losses and flow characteristics upstream of the trash racks. They found that trash racks with vertically angled streamwise bars were characterised by lower head losses than those caused by angled bars. However, angled bars are likely to be more effective at lowering fish impingement.

The angle of a trash rack can lower the velocity vector through the trash rack. This can reduce the risk of fish impingement on the rack itself or on any screens. Trash racks can be installed:

- perpendicular to the approach flow ($\alpha=\gamma=90°$);
- inclined with a vertical angle $\gamma < 90°$, or;
- angled with a horizontal angle $\alpha < 90°$ (Figure 7-29).

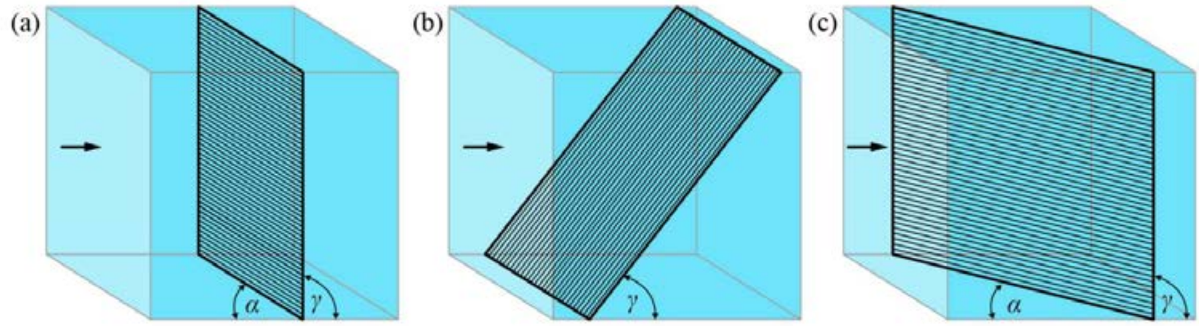


Figure 7-29: Rack layouts: a) perpendicular, b) inclined, and c) angled to the approach flow. α: horizontal approach flow angle, γ: rack inclination angle. Reproduced from Meister et al. (2022).

Angle

When designing new screens/racks, the horizontal approach flow angle (α) is selected to match the swimming capabilities and life stage of the relevant target fish species. For approach water velocities between 0.40 and 0.80 m s^{-1}, a rack angle of α = 20°–40° is suggested. To avoid impingement, international and New Zealand guidelines (Hickford et al. 2023) require the sweep velocity to be considerably greater than the approach velocity. A study by Raynal et al. (2013a) confirmed that for this to occur the trash racks must be sharply angled to α ≤45°. As angles become increasingly acute, water velocities along the screen/rack increase which helps to lead target fish towards a bypass (or some other fish intervention). This same study also recommended that to avoid impingement of silver eels that the approach velocity should not exceed 0.5 m s^{-1}.

Incline

In addition to potentially angling racks across a waterway to alter outcomes for fish, racks can also be inclined. Inclining a rack serves to get a fish to change their position in the water column and guide them to the downstream end of an inclined rack; the incline works in combination with ramp angle to get fish to move towards a bypass (or trap) location. Raynal et al. (2013b) examined the velocity profile of racks inclined at different angles (*β*) and found that to generate a sweep velocity at least twice as large as the through screen velocity, racks must be sharply inclined to *β* ≤25° to satisfy this recommendation. For *β* ≤25°, this study also showed that upstream mean velocities up to 1 m s^{-1} satisfy the recommendation on the normal velocity ($V_n \leq 0.5$ m s^{-1}) to avoid impingement of silver eels on the rack (Raynal et al. 2013a; Raynal et al. 2013b).

Bar shape/profile

Conventional trash racks are generally vertical and perpendicular to the flow direction and the profile of the bars is rectangular. There are now multiple types of bar profiles available including modified bar racks, curved bar racks, droplet, tadpole, etc., These bar profiles are proposed by manufacturers to lower head losses and are also increasingly being used in the design of structures to facilitate fish passage. New design materials can markedly reduce head losses and thus more fish-friendly screen designs may be possible that result in no net head loss for asset owners. For example, hydrodynamically shaped bars can generate head losses approximately 40% lower than the rectangular plate bars (Raynal et al. 2013a; Raynal et al. 2013b).

Velocity

Approach velocity is another important parameter to be considered when optimising trash rack design for fish exclusion. Approach velocity is the flow perpendicular to the front of a fish exclusion structure (Schwevers and Adam 2020). The issue with high approach velocity at the bar rack is the risk of fish impingement on the rack; fish get stuck/impinged on the rack and do not have the ability to escape off the rack and swim towards a bypass (if present). Reducing the approach water velocity towards the rack is critical to enable angles/inclines to be effectively used to guide the fish towards an escape opening/bypass. Present-day designs for water infrastructure where fish screening is required (e.g., irrigation intakes) also need to consider sweep velocity, to ensure fish are not impinged (i.e., trapped) against a screen by ensuring a velocity vector that moves them along a screen, usually towards a bypass. Generating a sweep velocity often requires modification of existing infrastructure and this is made more difficult at large dams because of their depth and existing intakes being perpendicular to the flow (i.e., there is presently no sweep velocity at these dams).

Trash racks as guidance devices

Trash racks are increasingly used in Europe at medium-sized hydro-schemes as fish guidance structures. The working principle of all fish guidance structures is as follows: the bars create highly turbulent flow zones, flow separations, and changes in water velocities and directions so that fish can sense them, react with behavioural avoidance and hence are guided to a bypass (or alternative escape route) by the velocity component parallel to the rack (Amaral et al. 2003; Albayrak et al. 2020). The efficiency of a bypass for downstream fish passage is strongly dictated by the hydraulic conditions at the entrance of the structure, and these conditions can vary with the design of the associated trash racks (Szabo-Meszaros et al. 2018). For fish guidance towards a bypass, it is now recommended to incline or angle trash racks. In the case of inclined trash racks, the size and the number of transversal elements (spacers between bars and girders) in more modern trash racks have increased to support the rack correctly (Lemkecher et al. 2021). Boes et al. (2016) indicated that trash racks with horizontal bars combined with a bypass can be a preferable solution for fish protection at smaller hydro-power plants, while trash racks with vertical bars can be an alternative for larger schemes.

7.4.7 Screening

National guidelines on screening design for New Zealand, as outlined in Hickford et al. (2023), provides a comprehensive set of criteria for designing effective water intake structures and associated fish screens. Following these principles, (Hickford et al. 2023) physical barriers, such as screens are used in conjunction with bypass systems to facilitate downstream fish passage. Fish screens protect fish by blocking the migration corridor towards turbines without causing damage to the fish species. The flow characteristics that are generated by the placement of the screen and the physical parameters of the screen itself then help to guide fishes and transfer them to a bypass system (Keunecke et al. 2020), Figure 7-30). Water intake structures are used throughout New Zealand to supply irrigation, hydro-electric generation, drinking water and industrial need. Intake structures are used to divert water from a river or lake. Here, the extraction of water may require pumping via an intake incorporating a sump, but many intakes are gravity-fed diversions. The fundamental purpose of a fish screen at a water intake is to ensure safe passage for all fishes around, or through, the intake structure within or back to the source river. The screening material is only one part of this process. It is also important that the intake design allows for, and incorporates, known fish behaviours to protect the fish community.

Water intakes, and associated fish screens, must be designed to minimise or eliminate the possibility of fish being damaged or removed from the waterway. Evaluating what constitutes an effectively screened water intake design, for fish, is often based on a suite of key criteria (Hickford et al. 2023) such as:

- the intake being located to minimises the exposure to fish and the distance from the waterway,
- providing appropriate intake (through screen) velocity and sweep velocity,
- having a bypass that prevents entrainment and impingement,
- ensuring connectivity between the constructed bypass and the mainstem of the waterway,
- using effective (i.e., gap openings that are small enough to exclude fish) and durable screening material,
- ensuring the intake can be maintained, effective and connected to the river, and
- ensuring that either the water intake and fish screen does not impede upstream passage during all flows, or that the bypass outlet impedes fish passage into the bypass keeping fish in the natural waterway.

National screening guidelines and design considerations are applicable to intake sizes up to 10 $m^3 s^{-1}$ surface and 500 $L s^{-1}$ pumped (Hickford et al. 2023).

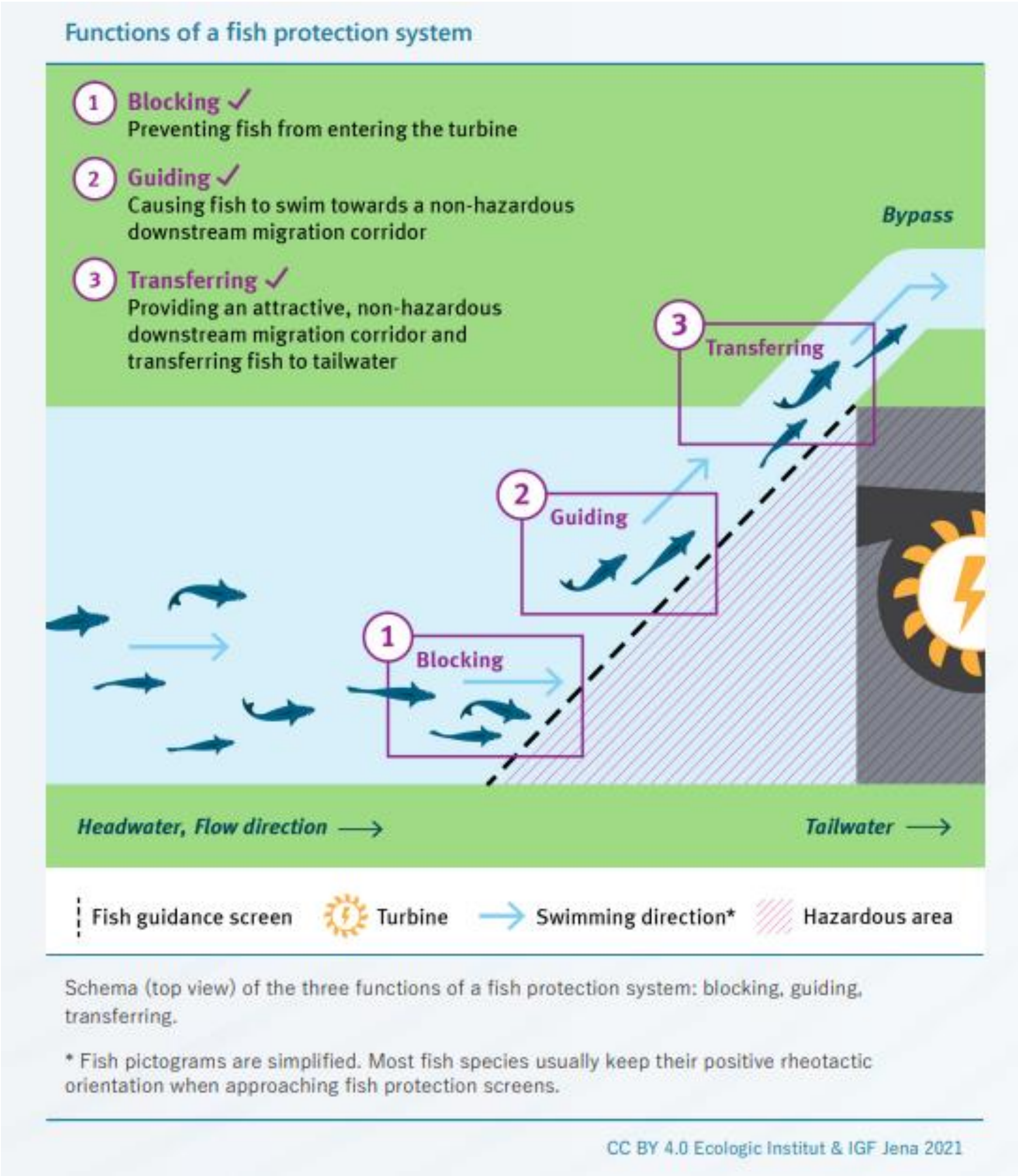
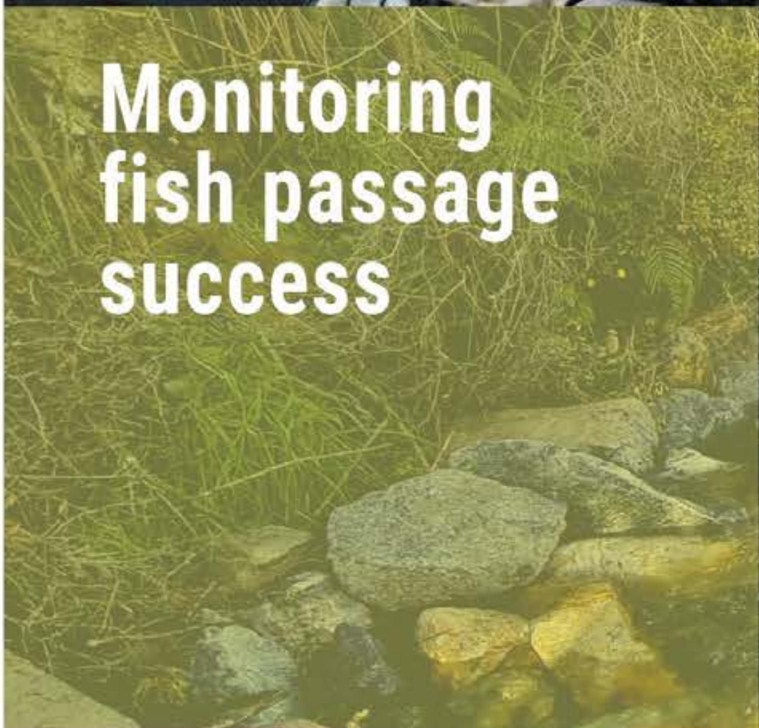


Figure 7-30: Conceptual illustration of screen guiding fish to bypass system (from Keunecke et al. (2020)).

Monitoring fish passage success



8 Monitoring fish passage success

Whether it is a new structure being installed or remediation of an existing structure, planning and implementing an effective monitoring and maintenance programme *before* the works is undertaken is an essential part of the project. Even when best practice guidelines are followed for the design or remediation of an instream structure, a well-designed monitoring and maintenance programme is essential to ensure the structure remains fit-for-purpose and meets the means objectives and performance standards (Section 3). Furthermore, evaluating the performance of a structure or fish pass can inform the level of mitigation that might be required to overcome poor passage efficiency at a structure. Well-designed monitoring programmes also help to increase knowledge of the function of different fish passage solutions and inform future improvements in design.

Monitoring is the only way to understand how well a structure is working and to ensure that any reduction in fish passage caused by a structure is not adversely impacting upstream communities and that environmental outcomes are being achieved. It is particularly important to understand how well a structure is functioning in situations such as:

- High value fish communities or ecosystems are present, or expected to be present, upstream of the structure.
- Unproven designs are being used.
- Proven designs are being used in novel situations.
- Retrofit solutions form only one component of an instream structure.
- Multiple structures exist within a waterway causing cumulative effects.
- Selective barriers are being used to manage the movement of undesirable species.

To assist in providing robust monitoring approaches for assessing the effectiveness of fish passage remediation, **a stand-alone manual "Guidelines for monitoring fish passage success at instream structures and fishways" has been developed by Baker et al. (2024a)**. The monitoring methods outlined in the manual by Baker et al. (2024a) will ensure that fish passage assessments are consistent across New Zealand, enabling a comparison of efficacy, and consequently ensuring that investment leads to the best possible outcomes for fish passage, catchment connectivity, and threatened species restoration. The following section provides a summary of key monitoring approaches and protocols, but it is anticipated that Baker et al. (2024a) is utilised alongside the present guideline document.

8.1 Identifying performance measures

Setting performance measures for fish pass design or remediation (Section 3) provides clear metrics for monitoring the effectiveness of the fishway/structure. Without setting clear performance measures a priori, there is a higher risk of drawing false conclusions from the data generated from effectiveness monitoring (Bunt et al. 2012; Mahlum et al. 2018). Physical and/or hydraulic objectives can often be utilised alongside biological monitoring of fish passage past instream structures. The following sections provide guidelines on biological monitoring approaches (Section 8.2) and physical and/or hydraulic monitoring approaches (Section 8.3) for fish passage projects.

8.2 Biological monitoring

In New Zealand, the key focus for fish passage remediation and new structure design is to provide unimpeded passage, particularly for smaller more vulnerable juvenile fishes. If remediation or installation of instream structures is carried out following methods in these guidelines, which will effectively promote the upstream passage of target fish species, effective downstream passage will likely be provided. **The following sections on biological monitoring focus on upstream passage of fish**.

To monitor the passage efficiency of downstream moving fish, biotelemetry methods are recommended (see Baker et al. (2024a)). In New Zealand, apart from lamprey, the main life stage of native fishes moving downstream are adults, which enables biotelemetry techniques to be utilised across a wider range of species. Biotelemetry is the recommended approach because downstream migrations are often undertaken on high flow events where netting and trapping is difficult or ineffective. In addition, the timing of downstream movements by adult fish are also less known, appear to be more variable and can occur in pulses. In contrast, the upstream movements of target species can generally be captured effectively (e.g., whitebait).

Figure 8-1 provides guidelines on identifying the desired biological performance measure for any structure and, subsequently, the most appropriate monitoring approach and survey method for evaluating those performance measures. Where other performance measures are identified, it is important to carefully match the monitoring approach and survey methods following a similar framework and to provide a defensible and transparent justification for the approach taken.

Multiple performance measures may require that monitoring approaches and/or multiple survey methods be utilised. Pairing BACI surveys with mark-and-recapture trials will provide the most robust assessment of passage efficacy for an instream structure. This would be the recommended approach for initially ensuring any new instream structure or remediation is fit-for-purpose and their effective operating range. Once sufficient evidence is available to have confidence in the effectiveness of solutions and the circumstances under which they are suitable, the need for comprehensive monitoring may be reduced.

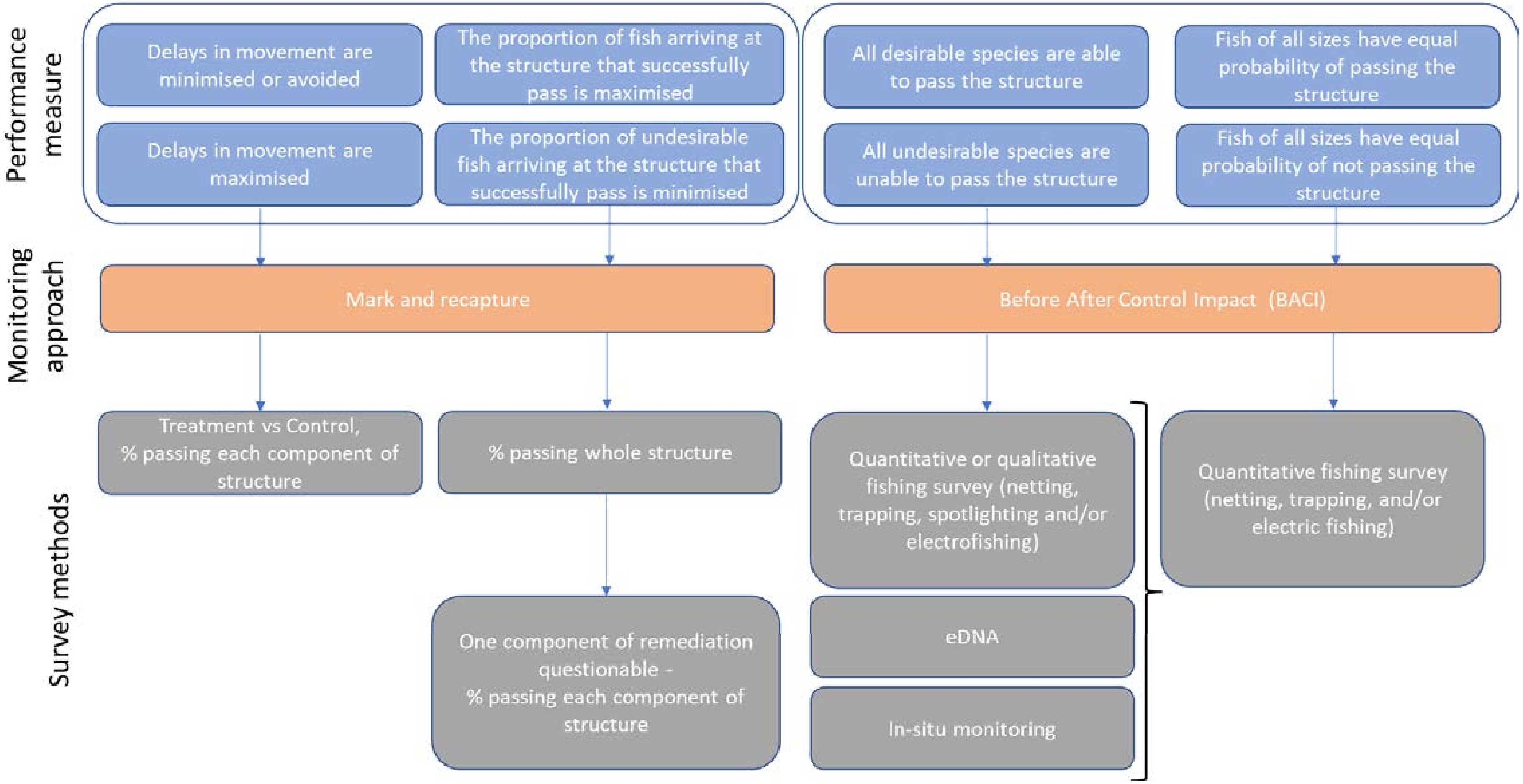


Figure 8-1: Identifying appropriate survey methods based on a priori performance measures.

8.2.1 Biological monitoring included in this manual

A range of monitoring approaches are available but based on the small body size of New Zealand's migratory fish, the two approaches recommended for evaluating upstream fish passage success are: a before-after-control-impact (BACI) survey, and/or an in-situ mark-and-recapture study. As such, this section focuses on BACI surveys and mark-and-recapture studies, which have the widest applicability for monitoring upstream fish passage with New Zealand species. The main benefits and drawbacks of a range of approaches are outlined in Table 8-1.

Table 8-1: The main benefits and drawbacks of before after control impact (BACI) and mark-and-recapture monitoring approaches.

Monitoring	Benefits	Drawbacks
BACI survey (e.g., electrofishing or netting surveys)	Documents changes to fish communities upstream of the remediated structure following intervention (e.g., structure removal or installation).	Can take several years to determine if the remediation is effective (e.g., recruitment of diadromous species can be variable between years).
	Minimises handling and stress to fish species.	If the retrofit is unsuccessful in promoting fish passage no information is provided on which component of the remediated structure is still problematic.
		Does not provide any indication of the proportion of fish successfully passing the structure.
Mark & recapture study (e.g., stain and release)	Can be used to test different components of an instream structure independently and collectively.	Fish are subjected to handling and stress, which may affect passage success.
	Immediate results on the effectiveness of the solution.	Does not document changes in upstream fish communities.
	Provides an estimate of passage efficiency.	May require permits from MPI, DOC or Fish and Game for the transfer and release of fish.
		Lack of ability to capture test fish downstream can limit use.

8.2.2 Other monitoring approaches

Other automated methods (e.g., biotelemetry) can also be utilised, but they generally require a higher investment in resources and have severe limitations in monitoring small-bodied fish with a slim morphology (i.e., juvenile galaxiids) that are often the focus of fish passage monitoring. Simpler methods such as visual checks can also be used but can be subject to observer bias and a lack of reproducibility. A brief overview of the main automated approaches utilised globally and their applicability to fish passage monitoring in New Zealand is provided below with advantages and drawbacks outlined in Table 8-2.

Table 8-2: The main benefits and drawbacks of various automated or visual monitoring approaches.

Monitoring	Benefits	Drawbacks
Biotelemetry (e.g., PIT, acoustic and radio tagging)	Timing and location of fish movements and behaviour can be captured.	Tags too big for some species and/or life stages and may alter behaviour.
	Remote data capture possible.	Battery life of tags may not be sufficient.
	Passage efficiency can be estimated	Tags and antennae can be relatively expensive.
Fish counters	Minimises handling of fish.	Does not document passage failure.
	Can be low cost.	Does not document changes in upstream communities.
		Does not accurately identify species.
Video and acoustic cameras (e.g., ARIS, DIDSON)	Avoids handling of fish.	Video processing can be laborious. AI technology currently unable to reliably automate fish counts, especially for small-bodied fishes.
	Can be relatively low cost.	Ineffective in water with poor visibility (video cameras).
	Can provide semi-automated monitoring of target species.	Generally restricted to enclosed areas and does not document changes in upstream communities.
		Does not accurately identify similar species, particularly when small, i.e., discriminate between īnanga and climbing galaxiid as whitebait.
Visual checks	Quick and cost-effective means of identifying potential problems.	Ineffective at quantifying passage success rates.
		Does not document changes in upstream communities.
		Ineffective in water with poor visibility.
		Does not accurately identify similar species, particularly when small, i.e., discriminate between īnanga and climbing galaxiid as whitebait.

Biotelemetry technology

Biotelemetry has been extensively used for monitoring fish passage worldwide (Roussel et al. 2000; Cucherousset et al. 2005; Aymes & Rives 2009; Baker et al. 2017). Biotelemetry technology requires fish to be tagged with either active or passive tags that can be internally implanted or externally attached. There are three main technologies utilised: Passive Integrated Transponder (PIT) telemetry, radio telemetry, and acoustic telemetry.

PIT tags have no battery and last indefinitely, which enables the tag to be physically smaller in size. Presently PIT tags are available in 8 mm and 12 mm sizes for full and half duplex, respectively. Radio and acoustic telemetry tags contain an internal battery and the size of the tag generally relates to the length of operation. Radio telemetry is effective in shallow freshwater environments whereas acoustic transmitters are more effective in deep water environments (Cooke et al. 2013). While radio telemetry cannot track fish accurately in 3D, acoustic telemetry has the capability of determining 3D positions with high accuracy and temporal resolution. For this reason, acoustic telemetry is often used to examine fish behaviour at high head dams (Nielsen and Szabo-Meszaros 2022).

The greatest limitations of using biotelemetry are the cost of the monitoring equipment and the size of the tag that can be attached to or implanted in individuals. The general rule of thumb is that the tag size represents no more than 2% of the individual's wet weight (Jepsen et al. 2003), although recent studies suggest this is a conservative limit for some species (Jepsen et al. 2003; Smircich and Kelly 2014; McKenna et al. 2021). However, the small bodied species and juvenile life stages that migrate upstream in New Zealand generally restricts the applicability of biotelemetry to monitoring adult lamprey or downstream tuna migrations at high head dams or within technical fishways. For guidelines on using biotelemetry to monitor fish passage success refer to Baker et al. (2024a).

Fish counters

A range of automated fish counters are available commercially that operate using either resistivity between the water and body of the fish or infrared beams. The VAKI Riverwatcher is one of the more widely used electronic fish counters that measures the size and shape of fish that pass through an infrared scanner (Jones and O'Connor 2017). Major disadvantages of fish counters are that they cannot discriminate between species other than by their size, and their accuracy is negatively affected by visibility, fish size, and speed of travel. For example, laboratory testing of the Riverwatcher found it underestimated counts of Silver perch (*Bidyanus bidyanus*; size range: 345–498 mm), a slow swimming species, by 56–84% at moderate migration rates (12 fish h^{-1}; Baumgartner et al. (2012)). Based on current limitations, fish counters are not recommended for monitoring upstream migrating fish through fishways or past instream structures in New Zealand.

Acoustic technology

Acoustic technology or sonar can collect real-time data on fish moving past or through fishways using acoustic sampling from fixed transducers. The sonar systems commercially available have high resolution and fast frame rates. Currently, the Adaptive Resolution Imaging Sonar (ARIS) camera is the most technologically advanced acoustic camera commercially available. The ARIS can capture details as small as a few millimetres, and view targets at a range of up to 40 m. Alternative acoustic technology includes the Dual-Frequency Identification Sonar (DIDSON) which uses sound to produce images of fish at ranges up to 15 m in high-frequency mode (1.8 MHz) and up to 40 m in low-frequency mode (1.2 MHz). However, image clarity for small fish (e.g., juvenile galaxiids) will be at a lower resolution relative to ARIS technology. In the past three decades, this technology has been extensively used at hydroelectric dams to study the approach and passage behaviour of upstream and downstream migrating fish (Nielsen and Szabo-Meszaros 2022). In New Zealand, the DIDSON has been used successfully to monitor rainbow trout and Chinook salmon movements at the Level Plain irrigation screen.[19] The disadvantages of acoustic technology are the intensive data processing due to the continuous surveillance, and the high capital investment. In addition, it is unlikely that sonar will accurately identify the five whitebait species and smelt within upstream moving shoals of juvenile fish. Their accuracy for enumerating small-bodied (<100 mm TL) fishes, particularly those that shoal, also remains subject to relatively high error rates (Wei et al. 2022). For these reasons, acoustic technology is recommended for use primarily in monitoring fish passage at large head dams and/or for larger fish species/life stages.

Video surveillance

Video surveillance provides another tool to monitor fish passage success and Artificial Intelligence (AI) using deep-learning and machine-learning techniques is emerging that filters video recordings to automate species detection (e.g. Kandimalla et al. 2022; Magaju et al. 2023). In time, automated video surveillance could reliably be used to identify different species, collect abundance information, passage rates, and the size structure of each species successfully moving through a fishway or past an instream structure. Currently, this technology is not fully developed for any fish species and generally performs poorly for small bodied (<150 mm) species/life stages (Egg et al. 2018). Consequently, video surveillance is not yet recommended for use in accurately monitoring upstream fish passage success in New Zealand.

8.2.3 Survey methods

This section focuses on selecting appropriate methodologies for BACI surveys and mark-and-recapture studies at a given structure. For detailed descriptions of the protocols refer to Section 8.2.4.

BACI Methods

Where the performance measure relates to the effects of improved connectivity on upstream fish communities, species richness at a site, and ensuring all size classes are represented within populations, the recommended long-term approach is to utilise a before-after-control-impact (BACI) survey design. This is where fish surveys are undertaken downstream (control) and upstream (impact) of the structure (assuming the focus is on upstream migration), before and after remediation is carried out. Before and After sampling will determine how the installation of a structure or structure remediation changed the fish community through time relative to its previous condition. Control and Impact sampling will allow effects of the structure to be discerned from natural variability, stochastic events, and underlying trends in fish populations in the wider area. The BACI survey design is widely used for environmental impact assessments. The main survey methods recommended for carrying out BACI surveys are:

- A physical fishing survey (qualitative or quantitative).
- Environmental DNA (eDNA) sampling.
- In-situ monitoring.

Regardless of which method is utilised a well-designed and balanced BACI survey remains one of the best methods for assessing environmental effects (Smokorowski and Randall 2017). That is, adequate

[19] Levels Plain irrigation fish screen trial 6-9 December 2010 (irrigationnz.co.nz)

pre-data are collected to provide a robust baseline for measuring effects against, and the same method (and deployment time for in-situ monitoring) is used for the before, after, control and impact components of the monitoring. The following section outlines each method and helps identify the applicable target species for any given site. Once an appropriate method is selected, the recommended protocols are provided in Section 8.4 below.

Fishing Survey

A fishing survey is the recommended option for all BACI monitoring as it provides the most information about community structure upstream and downstream of the structure. Utilise Figure 8-2 to determine the target species and, subsequently, the appropriate fishing method for the site from

Table 8-3. If a combination of electrofishing, fyke nets and Gee minnows is required to effectively capture the target species and life stages, then using all recommended methods will maximise the information gained and provide the most robust data for measuring whether objectives have been met. For example, if īnanga (weak swimmer), common bullies (weak swimmer) and banded kōkopu (good climber) are the target species, then Gee minnow traps and fine mesh fyke nets would be the effective fishing methods. If redfin bully (weak climber) were also targeted, then electrofishing would be necessary as fyke netting can underestimate their abundance.

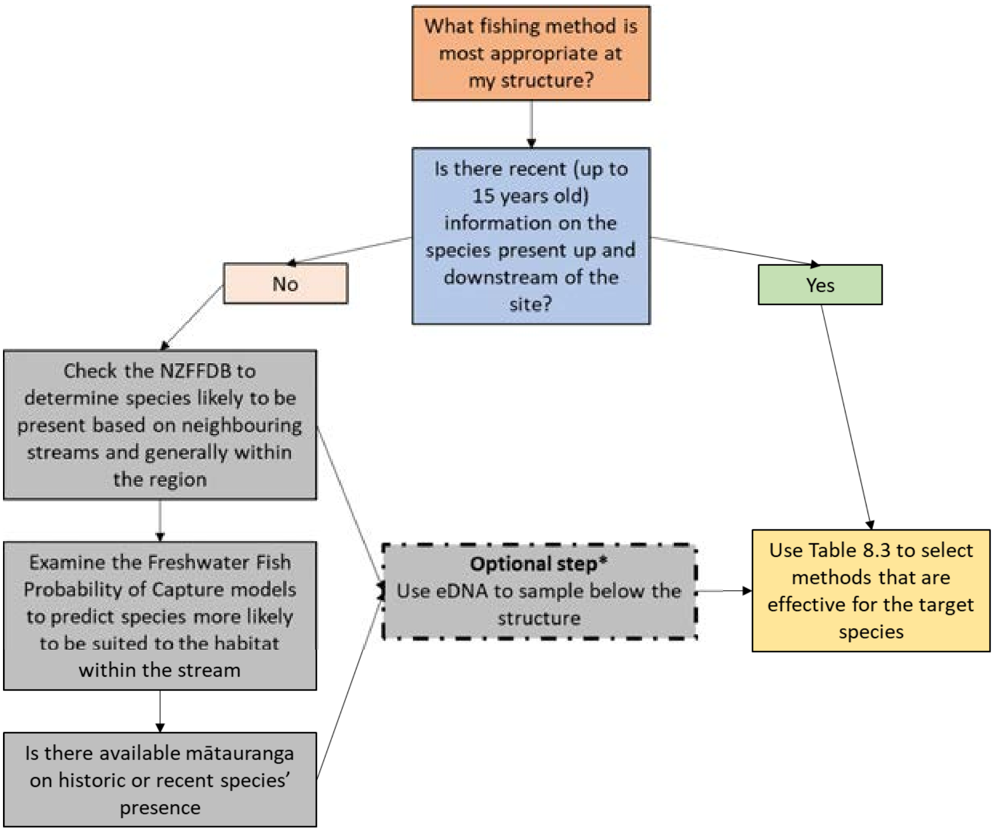


Figure 8-2: Guidelines on fishing method selection[20]. Desktop tools to determine appropriate target species. Abbreviations: NZFFD, New Zealand Freshwater Fish Database. * As eDNA detects species present upstream of the sampling location, Melchior & Baker (2023) recommend surveying 300–500 m below the instream structure to effectively sample DNA from fish resident within the waterway as well as those congregating below, or delayed by, the structure.

[20] Use the following link to download all records held in the NZFFD (https://nzffdms.niwa.co.nz/search). The NZ species DB (downloadable from Jowett Consulting - NZ Species DB) is a useful tool for visualising records from the NZFFD. The probability of capture models by Crow et al. (2014) can be accessed through NZ River Maps.

Table 8-3: Effective methods for target fish species. The species included in the table below are the main species that would be targeted for monitoring fish passage past instream structures along with some key resident non-migratory species. For swimming and climbing modes of movement, the strength of their ability is indicated in brackets for those species where data exists. Abbreviations: C, capable of climbing; S, must swim past obstacles; E, effective method; U, can be used but underestimates abundances; FF, fine mesh fyke net; CF, coarse mesh fyke net; GM, Gee minnow trap; G, gill net.

Species	Life stage	Swimming/ climbing ability	Electro-fishing*	Trapping/ netting	Spotlighting	Seining
Native species						
Longfin & shortfin eels	Elver	C (strong)	E	U (FF)		
	Adult	S	U	E (FF & CF)		
Īnanga	All	S (weak)	U	E (GM), U (FF)ᴪ		
Smelt	All	S (moderate)	U	U (FF)ᴪ		E
Banded kōkopu	All	C (good)	U	E (FF)	E	
Giant kōkopu	All	C	U	E (FF)	E	
Shortjaw kōkopu	All	C	U	E (FF)	E	
Kōaro	All	C (strong)	E	U (FF)	U	
Redfin bully	All	C (weak)	E	U (FF & GM)	U	
Common bully	All	S (weak)	E	E (GM), U (FF)ᴪ	U	
Bluegill bully	All	S	E	U (FF & GM)	U	
Torrentfish	All	C	E	U (FF & GM)		
Lamprey	Juvenile	S	E			
	Adult	C (strong)		E (FF)	E	
Mullet	All	S		E (G)		
Non-migratory bullies	All	S (weak)	E	E (GM)		
Non-migratory galaxiids	All	S	E		E	
Introduced species						
Salmonids	All	S (strong)	E*	E (G)		
Catfish	All	S		E (FF), U (G)		
Perch	All	S	E*	E (G)		
Koi/amur carp	All	S		U (G)**		
Goldfish	All	S		E (G)		
Rudd	All	S		E (G)		

*wadable stream only. For non-wadable streams an alternative method will be required such as boat electrofishing or netting.

**in lakes and deep water, boat electrofishing is likely the most effective method of capture.

ᴪCan be effectively captured in fyke nets when eel excluders are utilised. Without preventing eel entry, the presence of large predatory eels can influence the behaviour and capture efficiencies of these prey species.

Quantitative fishing methods

If the performance measure requires understanding species richness and a robust assessment of population structure and abundance upstream and downstream of the structure, quantitative fishing methods are required. Here, fyke netting, Gee minnow trapping, gill netting and multi-pass depletion electrofishing are the appropriate survey methods with selection contingent upon target species and habitat at the site.

Qualitative fishing methods

If the performance measure requires understanding species richness and relative abundance of species upstream and downstream of the structure, but a robust assessment of population structure and abundance is unnecessary, then qualitative fishing methods can be employed. Here, fyke netting, Gee minnow trapping, gill netting are still appropriate if they are the effective method for the target species. However, spotlighting and the standardised electrofishing protocol can be implemented.

The standardised electrofishing method (David and Hamer 2010; Joy et al. 2013) utilises a single pass of 150 m reach and provides the relative abundance of fish species. The method was designed primarily for detecting maximum reach scale diversity (species richness) across a variety of stream types ensuring all instream habitats are sampled, rather than an assessment of species densities over a shorter stream length.

Spotlighting is an effective method for detecting non-migratory and large galaxiids, however, we do not recommend spotlighting as a quantitative method for assessing the effectiveness of fish passage remediation. This is because the observer’s skill level strongly influences the effectiveness of the spotlighting technique. In addition, walking upstream creates vibrations detectable by the fishes’ lateral line and/or vestibular systems, which can cause a proportion of the nocturnal fish being targeted to dart or move to cover. This predator avoidance response can vary within and between fish species contingent upon environmental factors (e.g., lunar cycle), and between stream types depending upon the riparian margins and ease with which the stream banks can be navigated. Should spotlighting be undertaken as a survey method, then pairing it with fyke netting, electrofishing or eDNA is recommended to reduce the potential impacts of observer bias.

Environmental DNA (eDNA)

eDNA is a relatively new technology that has yet to be validated for use in monitoring fish passage efficacy at instream structures (Melchior and Baker 2023). However, for structures that likely form a severe impediment to fish passage, eDNA sampling could be useful for examining differences in species diversity downstream and upstream of a structure. eDNA monitoring would not be able to determine if certain size classes of fish species were restricted by the instream structure, nor is there currently enough information to translate DNA reads into an accurate measure of abundance or density. Consequently, eDNA is not a quantitative assessment, but it could provide evidence for presence/absence of weak-swimming fish such as īnanga before and after remediation of an obstacle. It can also help identify if undesirable fish species start penetrating past the instream structure after remediation. As eDNA reads do not correlate directly to fish abundance, it should be noted that if the instream structure is an impediment but passable intermittently, eDNA sampling may not conclusively determine a change in species richness after remediation.

In-situ monitoring

If the performance measure requires understanding the species and size classes passing the structure, but quantitative information on the size structure of populations upstream and downstream of the structure is not a focus, then in-situ monitoring by setting traps at the structure inlet and outlet can be undertaken. This method is only effective if the entire stream immediately upstream of the structure and at the outlet can be completely blocked off. In this regard, it is most suited for examining movements of fish through low head structures such as culverts. In general, rigid A-Frame or whitebait traps, or double winged fyke nets have been successfully utilised at low head structures for mark-and-recapture studies (e.g. Franklin and Bartels 2012; Amtstaetter et al. 2017; Jones and O'Connor 2017) and would be the recommended option for this short-term in-situ monitoring. It is important that the mesh size of the nets used ensures capture of the smallest target fish.

Setting traps at the entrance and exit of the structure can determine if the fish population (diversity, abundance, and size classes) entering the structure is comparable to those successfully exiting the structure (e.g. Bice et al. 2017; Jones and O'Connor 2017). In Australia, trapping the entrance and exit of a fishway is generally the first method of assessment utilised (Jones and O'Connor 2017). Here, a qualitative assessment of the effect of the fishway can generally be attributed to any differences in the fish population captured at the entrance and the exit. Sampling the entrance and exit of the structure before and after remediation provides a proxy for a control and impact for passage success. The control is the entrance of the structure, which typically represents the fish population downstream of the barrier, while the exit is the treatment, or the impact for the fish trying to pass the structure (Jones and O'Connor 2017). This method does not represent a true control as it does not sample downstream away from the influence of the structure. In addition, it will only capture fish actively migrating through the structure, so the monitoring period may not represent all species and life stages attempting passage.

Setting traps simultaneously upstream and downstream must be avoided as the capture of fishes in the downstream trap influences their movements, which can have effects on motivation, behaviour, and swimming ability if physical damage is incurred during the trapping and handling process. These unquantified impacts on fish movements can also vary between species. For example, fragile species such as smelt die from brief exposure to air and stress due to confinement.

In some situations, trapping at the entrance to a structure may not be feasible. Here, setting the traps upstream of the obstacle still provides valuable information on the fish population (diversity, abundance, and size classes) passing the structure over the monitoring period. This can be directly compared to that observed before remediation of the structure. However, by setting traps only at the inlet (i.e., upstream end) of the structure, this Before-After (BA) method of monitoring does not determine if all fish moving upstream are able to successfully pass the structure as it doesn't document those that fail. To determine if the structure is restricting fish passage to stronger individuals, trapping should be paired with physical fishing survey methods outlined above or mark-and-recapture surveys.

Mark-and-recapture methods

Mark-and-recapture studies allow quantification of the proportion of fish that pass a structure (i.e., passage efficiency). This information is valuable as it allows the relative performance of different structure types or fish passage solutions in each situation to be established. This is essential to optimising fish passage outcomes at a site because the best solution for optimising fish passage can be more readily identified.

A mark-and-recapture study is recommended to:

- establish the performance and operating range of a fish passage solution that is to be installed across a range of sites,
- quantify the effectiveness of a solution that has not been demonstrated in practice, or
- to evaluate the relative influence of different components of a structure on overall fish passage success. For example, remediation of perched culverts commonly entails retrofitting a fish pass to the culvert outlet, yet the culvert barrel or transition from the fish pass to inside the culvert may still represent an impediment or barrier to certain fish species.

Because this type of study requires the stream to be barricaded at the top and bottom of the test reach, it is difficult to carry out in large non-wadable rivers and streams, or streams with high discharges and water velocities. For larger, high flow systems a BACI survey using nets and traps may be more applicable.

The trial design is dependent on the structure type and layout, and the performance measure being assessed. If the performance measure relates to understanding what proportion of fish arriving at the structure are successfully passing, then assessing passage across all components of a structure is important (e.g., up a fish ramp and through a culvert). However, assessing passage rates across individual components of the structure provides greater insight into the main constraints on fish movements across the structure and can be used to evaluate the effectiveness of specific mitigation actions that may target individual components of the structure (e.g., passage success over fish ramp compared to passage success through the culvert). If only one component of a structure is remediated, or the implemented solution has not been robustly tested, then testing each component separately is recommended.

If the performance measure is to determine if the structure delays or restricts fish passage, then having a control for testing the structure alongside is imperative. Here, a true control involves sampling either downstream or upstream of the instream structure to determine the migration rate of fish in the absence of an impediment.

Because of the small size of New Zealand’s freshwater fish species during their migratory stage there are limited options available for marking individuals. Based on laboratory and field studies, staining fish with Rhodamine B or Bismarck Brown is recommended over Visual Implant Elastomers (VIE) tagging, or other types of individual marking (e.g., coded tags, fin clips) where anaesthesia and handling is required. For example, laboratory studies utilising īnanga stained with Rhodamine B have found no reduction in swimming performance compared with unmarked control īnanga ($p = 0.68$) (Franklin et al. 2024). In contrast, īnanga with VIE tags swam at less than half the speed of unmarked control fish in critical swimming speed tests ($p = 0.005$) (Franklin et al. 2024). In addition, mark-and-recapture investigations at Bankwood Stream culvert, Hamilton (Franklin et al. 2024) found no significant difference in the number of īnanga stained with Rhodamine B and unmarked fish successfully passing the culvert ($p = 0.501$). Collectively, these data indicate that the staining marking procedure does not unduly influence the behaviour or passage ability of īnanga compared to fish that were not subjected to the marking procedure.

If all fish can be effectively removed from the barricaded test area and are unlikely to be able to migrate back inside the nets, then staining or tagging fish may not be essential for recapture trials. Unmarked fish can be released downstream of the structure and monitored for passage success.

8.2.4 Protocols

BACI

Fishing survey

A minimum of one survey reach upstream and one survey reach downstream of the structure is required for a BACI survey. As far as practicable, the two survey reaches should have similar habitat types and be of a similar size. This helps to minimise the potential influence of habitat availability and stream size on differences in fish communities between the control and impact sites. Consideration should also be given to locating the downstream survey reach slightly away from the immediate vicinity of the structure. Upstream migrant fish may aggregate immediately downstream of a barrier as they attempt to move upstream, so if the downstream survey reach includes these aggregations, fish population estimates can be biased and over-exaggerate the relative differences in fish community composition. To ensure the test site can be effectively monitored, walk the site during daylight hours to ensure there are no additional fish passage barriers, adjoining tributaries or other factors that would deem the reach unsuitable for monitoring.

When undertaking sampling as part of a BACI survey, regardless of what method is used, it is critical to ensure that data are collected in a consistent, standardised, and reproducible way. This means that for both the control and impact reaches, and before and after remediation:

- Sampling is carried out using the same method for each survey.
- The same sites are used for each survey.
- Sampling effort is equivalent between reaches and surveys (i.e., the same area is fished).
- Sampling is carried out under similar conditions (e.g., similar flows) for the before and after surveys.
- Sampling equipment (nets & traps) are the same for every reach within and between surveys (i.e., the before, after, control and impact reaches)
- Sampling upstream and downstream of the structure is carried out on the same day and the before and after surveys are carried out at the same time of year (i.e., within the same calendar month).

Table 8-4 outlines the protocols for each of the recommended survey methods. Use the guidelines in Sections 8.1 and 8.3 above to determine the appropriate monitoring approach and survey method based on the performance measure(s) at the site. Additional information on using each technique is detailed below.

Table 8-4: Reach size and fishing details for each survey method. The recommended reach lengths are the minimum required. Larger reaches can be utilised if preferrable.

Method	No. nets/traps	Baited	Reach length (m)	General guidelines
Fyke nets (coarse and fine)	6	No	200	Set fyke nets in deeper pools or slow-moving water. For single leader nets, if possible, bisect the stream setting the leader hard against one stream bank and secure the cod end across the stream against the opposite bank (see Figure 8-3). For double winged nets, if possible, span the width of the stream with the opening facing downstream.
Gee minnow trap	20	Yes	200	Bait with marmite or trout pellets. Set traps approximately 10 m apart. The exact location will be dictated by habitat availability, e.g., pools or slow-moving runs.
Electrofishing (multi-pass)	-	-	50	Set a stop net across the stream at the top and bottom of the reach. Fish all habitat in an upstream direction being careful to disturb as little sediment upstream of the fishing area as practicable. Remove all captured fish and repeat fishing of the reach until a 50% reduction from the previous pass is achieved in every species or 5 passes are undertaken.
Electrofishing (single pass)	-	-	150	Follow the protocols outlined in Baker et al. (2024a). Fish all habitat in an upstream direction being careful to disturb as little sediment upstream of the fishing area as practicable. Remove all captured fish and process at 15 m intervals (10 sub-reaches per site).
Gill nets	6	No	300	Set gill nets in the habitat utilised by target species. If possible, bisect the stream setting the net hard against one stream bank and secure across the stream against the opposite bank.
Spotlighting	-	-	200	Keep reach length and effort (time taken to sample the 200 m reach) consistent for all monitoring i.e., before, after, control and impact.

Nets & traps

Standardisation of sampling gear within and across monitoring programmes is important for obtaining robust data that can be compared spatially and temporally. In general, most agencies are utilising the same fyke nets (5 mm mesh stretched with eel excluders) and Gee minnow traps (3 mm mesh). Coarse mesh (12 mm stretch mesh) fyke nets are sometimes used in preference to fine mesh nets when only targeting eels. The fine mesh fyke nets and 3 mm Gee minnow traps have been rigorously field tested and will accurately document the population structure and size classes of fish successfully passing the instream structure. We recommend mesh sizes are, therefore, not increased over those previously stated as capturing all size classes of the target fish species forms one performance measure. It is also important to note that if equipment differs from that held by most regional councils and government agencies this restricts comparability of data and generalisation of results. Although size of the net opening, leader length and number, and other design attributes can differ between nets, the most important factor is consistency between reaches. That is, whatever fyke or minnow trap is used in the before monitoring must be used in both the upstream and downstream reaches in both the before and after monitoring. Changing net types within a BACI sampling programme will change catch efficiency and can lead to false conclusions being drawn.

Gill nets are effective at capturing pelagic fish species but are most frequently used in deep open water such as lakes. However, they can be effectively deployed in large rivers and wadable rivers to capture pelagic species. For example, gill nets may be used to examine the effectiveness of built barriers at preventing trout or pest species' access to upstream habitats. The capture efficiency of gill nets is strongly influenced by the characteristics of the net used, such as mesh size and filament diameter. There is no standard gill net type used in New Zealand freshwaters so comparability of results using this method is limited. It is important to remember that gill nets are a destructive fishing method, with captured fish having very high mortality rates. For invasive or nuisance exotic species gill net mortality is not an issue, but it should be considered when targeting mullet or sports fish. In addition, diving birds can frequently get caught in gill nets and suffer unintended mortality. In general, the type of gill net (panel or single) and the size of the mesh should be selected based on the target species and life stage(s). As stated above, the most important factor is utilising the same net for all survey reaches and sampling occasions within the BACI monitoring programme.

Survey protocol:

1. Select a reach upstream and downstream of the structure (minimum of 200 m or 300 m dependent upon net/trap type used).
2. Walk each reach to determine the suitable areas for net/trap deployment and record the GPS location of the top and bottom of each reach.
3. Actual set locations will be dictated by the presence of suitable habitat (e.g., pools and slow-moving runs). However, aim to spread the nets as evenly as practicable through each reach (Figure 8-3). That is, Gee minnow traps will be spaced approximately 10 m apart, with fyke nets spaced 30–35 m apart and gill nets approximately 50 m apart.

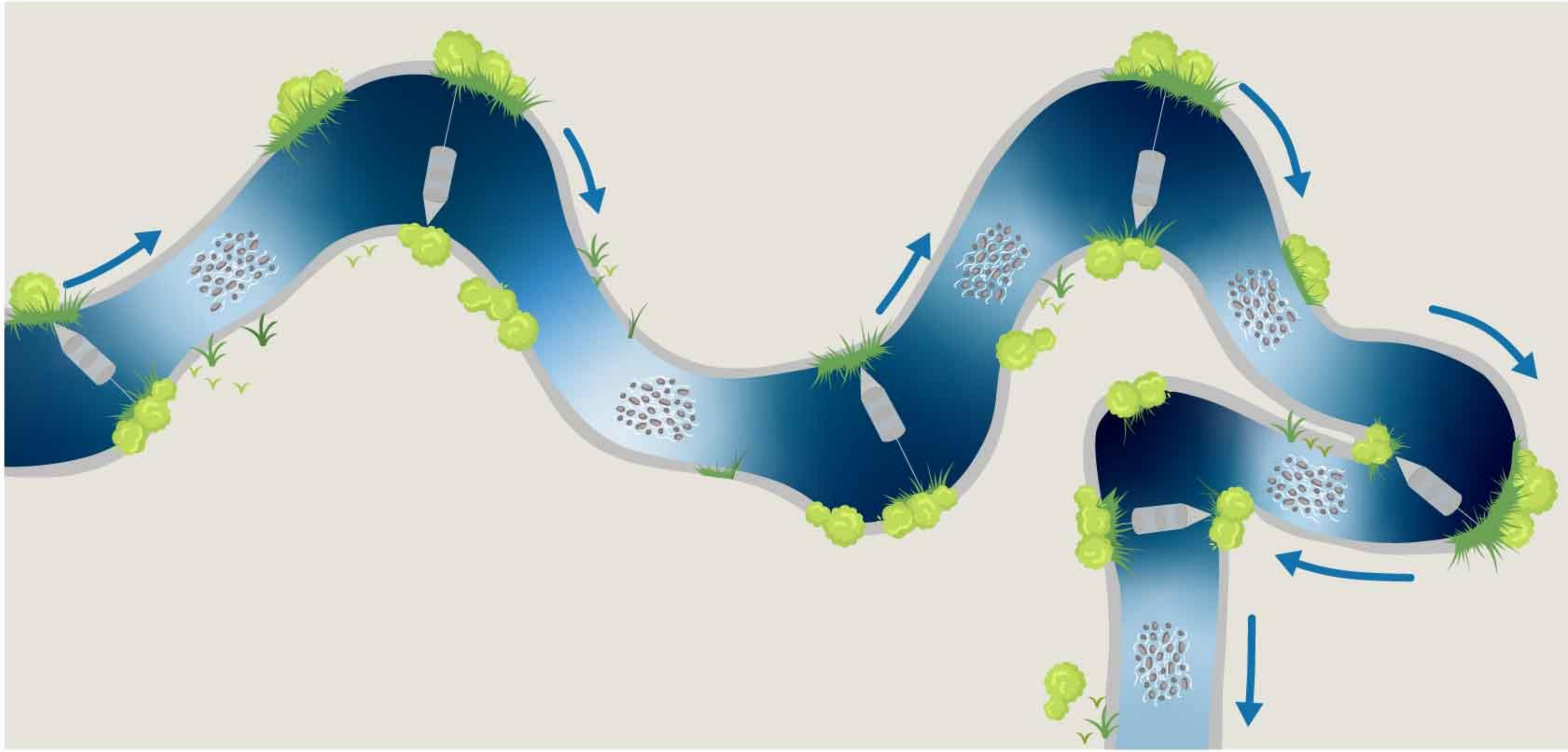


Figure 8-3: Conceptual layout of six fyke nets set within a hypothetical survey reach. Riffle sections are signified by the cobbles with pools indicated with deeper blue shading. Arrows indicate the direction of flow.

4. Setting nets/traps:

    4.1 *Fyke nets*. Set fyke nets in deeper pools, edges of pools, or slow-moving water. For single leader nets, where possible, bisect the stream setting the leader hard against one stream bank and secure the cod end across the stream against the opposite bank (Figure 8-4). In clean streams, net openings can face in any direction (upstream, downstream or perpendicular to the bank) as long as the leader is set hard against one stream margin. For streams with high debris loads, face the opening either perpendicular to the stream flow or in a downstream direction as otherwise debris can enter the net and clog the opening. For double winged nets, if possible, span the width of the stream with the opening facing downstream.

Figure 8-4: Fyke nets set in a large (A) and small (B) river.

    4.2 *Gee minnow traps*. Set Gee minnow traps upon the substrate in runs or pools with their long axis in-line with the flow (i.e., trap openings face directly upstream and downstream). This allows fish tracking the odour from the bait to easily enter the trap rather than needing to search for the opening. Tie each trap to bankside vegetation or a stake in the bank (Figure 8-5).

Figure 8-5: Gee minnow trap set in a pool of a small stream. The trap openings are parallel to the stream flow (i.e., facing upstream and downstream).

4.3 *Gill nets.* Target the feeding habitat utilised by the desired species. If possible, bisect the stream setting the net hard against one stream bank and secure across the stream against the opposite bank (Figure 8-6).

Figure 8-6: Panel gill net set across a stream at the head of a large pool.

5. If the stream is deemed to be degraded such that low dissolved oxygen could be present overnight (e.g., stream choked with macrophytes, sluggish to no flow and high summer temperatures), then fyke nets and Gee minnow traps should be set with an air gap at the top to enable captured fish to move to the water surface to extract oxygen from the air, if required.

6. Record the GPS coordinates of each fyke and gill net location. For Gee minnow traps, note the GPS coordinates at the top and bottom of each reach.

7. Set nets/traps in the afternoon and leave to fish overnight. If diving birds are present in the area, then caution is needed, and nets should be checked frequently during daylight hours and removed before dawn. For areas without diving birds, fyke and gill nets should be lifted the following morning. Gee minnow traps can be left for 24 hours to capture diurnal species. Record the set and lift times for each net/trap.

8. Process all fish captured to species. For each net/trap, record the length of the first 50 fish of each species, counting the rest of the individuals. It is important to accurately record the net/trap number each fish was captured from to provide a catch per unit effort where effort is defined as an individual fyke net, gill net or Gee minnow trap within each reach.

9. Record the wetted width and a minimum of three depths across the stream at equal intervals (e.g., every 20 m for a 200 m survey). This will allow the calculation of the area fished in m^2 and give an indication of water depths and change in flow between sampling. Utilise the standardised data sheet in Baker et al. (2024a), with one sheet used for each reach (upstream and downstream). Record GPS locations for the top and bottom of the upstream and downstream reaches and fill out NZFFD summary habitat assessment.

10. Data sheets can be custom made or follow the format documented in Baker et al. (2024a).

Electrofishing

For all electrofishing whether single or multi-pass, the following general protocol should be adhered to:

- Record the GPS coordinates at the top and bottom of each fishing reach.

- Choose your machine settings. This will be determined by the conductivity of the survey water and target species. Use fish response as an indicator of effective settings. Fish should be stunned and able to be captured within a hand net without swimming away, but with a quick recovery to swimming upright and balanced inside the holding bucket upon exiting the electric field. The recommended starting settings of 30 pps (pulse rate frequency in pulses per second) with a pulse width of 2 milliseconds (ms) are conservative and should not over shock large fish. If target fish are not effectively stunned increase the pulse frequency to 60 pps (pulses per second) and the pulse width to 3 ms.

- Use two or three people to fish the site, one operating the machine, one holding a pole (or push) net downstream of the fisher, and if possible, a third to hold the bucket and a hand-held dip net to help capture shocked fish.
- The fisher starts on the edge of either bank, fishing in a downstream direction towards the pole net covering between 1–2 linear metres of stream. If visibility is poor and/or fish are escaping, shorten the length of the fished area accordingly. If the stream is wider than a pole net width, the pole netter and the fisher then move horizontally across the stream one pole net width to fish the next section of water, continuing this protocol until reaching the opposite bank. In wider streams, and if capacity allows, multiple pole nets can be set and fished, or multiple fishers can be used, each with a pole netter.
- Once reaching the opposite bank, the pole netter and fisher move upstream so the pole netter is situated at the top of the area just fished and repeat the procedure continuing in an upstream direction and from bank to bank.
- Process all fish captured to species. Record the total length (nose to distal end of the caudal fin; mm) of the first 50 fish of each species, counting the rest of the individuals. Kōura and shrimp are counted rather than measured. Record exact numbers of kōura captured and record shrimp (*Paratya* sp.) into one of the following categories: 1–10, 11–100, 101–1000, 1000+.

Single pass

The standardised electrofishing protocol is a single pass survey without the use of stop nets at the top and bottom of the reach. As such, the results generated are the relative abundance of fish species, which is not equivalent to fish density and can only be used for a relative comparison of species diversity or richness at a site over time.

Survey protocol:

1. Utilise a 150 m reach at each site.
2. Use a hip chain during fishing or measure 10 × 15 m subreaches within the 150 m reach. Flagging tape can be set along the stream margins to delineate each 15 m subreach.
3. At the end of each 15 m sub-reach, process all fish captured.
4. Measure the wetted width of the stream at the end of each sub-reach.
5. Continue fishing until all sub-reaches are fished and fish are processed. Record the number of sub-reaches sampled on the collection form (use the form from Baker et al. (2024a)). Ideally, 10 sub-reaches will always be fished but if habitat is limited or other factors restrict the number of sub-reaches possible, check the appropriate circle, e.g., 5–9 subreaches, <5 subreaches.
6. Record the total shock time (elapsed time on the back of the fishing machine), the voltage used, along with the actual start and finish time for the total reach. This allows sampling effort to be calculated and compared between sites.

Utilise the data collection form provided in Baker et al. (2024a).

Multi-pass depletion fishing

If the objective is to quantify changes in fish numbers over time in response to changes to a structure, multi-pass depletion fishing is required to generate population estimates and true estimates of fish density. This allows a quantitative comparison of fish communities before and after remediation of the passage barrier within and between sites, and improved detection of population trends over time.

Survey protocol:

1. Utilise a 50 m reach at each site.
2. Set stop nets at the top and bottom of each reach before fishing.
3. Carry out multiple electrofishing passes until there is at least a 50% reduction in the catch of the main fish species compared with the previous pass or a maximum of five passes, whichever is reached first. Generally, three passes are the minimum necessary.
4. Fish and habitat information (e.g., fish lengths, wetted stream widths) should still be collected, but with five 10 m sub-reaches assessed instead of ten reaches.
5. Record the total shock time (elapsed time on the back of the fishing machine), the voltage used along with the actual start and finish time for the total reach. This allows sampling effort to be calculated and compared between sites and passes.

For three pass depletion fishing, population estimates for each species in the reach can then be calculated using the explicit approximation of the maximum likelihood formulae from Cowx (1983):

$$N_o = \left(6X^2 - 3XY - Y^2 + \left(Y \times \sqrt{(Y^2 + 6XY - 3X^2)}\right)\right) / \left(18 \times (X - Y)\right) \tag{19}$$

Where N_0 = population estimate, c_n = the number of fish captured in pass *n* and $X = 2c_1 + c_2$ and $Y = c_1 + c_2 + c_3$. Population estimates for multiple pass fishing surveys can also be calculated using the method of Zippin (1958) as executed in the removal function (http://www.rforge.net/FSA/) in R (http://www.R-project.org).

The density of each fish species in each section can then be calculated by dividing the population estimate by either the length of stream fished, to give the number of fish per linear metre of stream, or the stream area, to give the number of fish per metre square.

Spotlighting

Spotlighting surveys should be paired with fyke netting, electrofishing (after spotlighting undertaken) or eDNA to reduce observer bias. As such, to be consistent with fyke netting protocols (see Table 8-4) reach length should be at least 200 m. Ideal spotlighting conditions are a calm overcast night on a new moon when stream flows are low and the water is clear. Avoid nights where there is rain or strong winds that affect the water surface. Even small spots of rain can affect the clarity of the stream and result in biased data. Under a full moon and a bright night, some fish species have been noted to be easily spooked and more difficult to capture (Allibone 2013).

Survey protocol:

1. Delineate a 200 m reach upstream and downstream of the targeted instream structure. Wherever possible try and ensure similar meso-habitat is present both up and downstream, and that reaches avoid the outlet pool downstream of the structure. Measure the wetted width and minimum of three depths of the stream at 20 m intervals to give 10 width measurements and 10 sets of three water depths along each reach. This will allow the calculation of the area fished in m^2 and give an indication of water depths and change in flow between sampling. Reach length can be increased to accommodate site-specific factors; however, a minimum length of 200 m is recommended.

2. Ensure the water clarity is adequate for spotlighting. If streams are heavily tannin stained, have high levels of iron flocculant or suspended sediment, spotlighting is likely to be ineffective and netting or trapping should be carried out instead.

3. Utilise the standardised electrofishing and spotlighting data sheet in Baker et al. (2024a), with one sheet used for each reach (upstream and downstream). Record GPS locations for the top and bottom of the upstream and downstream reaches.

4. Prepare all equipment away from the stream to prevent noise and light affecting/spooking the fish.

5. Begin the spotlighting surveys around 45 minutes after sunset. Record the start time of each reach. Walk in an upstream direction. Walk on the stream bank if possible. If working in teams in wide streams, divide up the stream channel to ensure all the width is covered and quietly move upstream together. Here, walking within the stream will be necessary.

6. Shine the spotlight 1–2 m ahead and sweep from bank to bank. Do not scan the beam more than 4 m ahead to avoid spooking fish further upstream. As fish are sensitive to vibrations and noise, if you need to stop, stop beside a riffle where the chances of fish moving upstream is reduced.

7. Try to move at a slow but constant pace examining all habitats carefully for both pelagic and benthic fish species. Identify and count all fish you see in each of the 10 × 20 m subreaches. If fish are seen but can't be identified record them as "unknown" or identify them to the lowest taxonomic level possible such as "unidentified bully", "unidentified eel" and "unidentified kōkopu". If you can, capture fish to accurately record their length. If fish that cannot be captured can be conclusively identified, then record an estimate of their length.

8. If fish evade capture, turn all torches off and remain motionless for around 2 minutes. In some instances, fish will reemerge and a second chance at capture can be attempted. It is useful for each person to have a 30–50 watt spotlight and also a dim or red head lamp. This is because once a fish is seen it can be easier to catch without spooking with lower light levels.

9. Record the start time of each 20 m subreach and the time upon completion of the full 200 m reach. This allows total effort to be calculated along with identifying if any subreach was more difficult to navigate and required additional effort (time).

Each repeat spotlighting should be carried out within the same calendar month, at the same time of the night and sample the same 200 m reach. As far as practical, carry out repeat surveys under similar stream conditions and lunar phase. Try to keep the time taken to survey the reach the same between sampling occasions to keep fishing effort similar for each subsequent survey. Of all the methods described here, spotlighting is most susceptible to the influence of operator bias. Consequently, as far as practicable, the same observer(s) should be used for each survey.

Environmental DNA (eDNA)

Melchior and Baker (2023) have developed a living guideline document for using eDNA to sample lotic freshwater environments. Refer to Melchior and Baker (2023) for current and complete protocols on using eDNA to sample above and below fish passage barriers. Key protocols are outlined below.

To ensure that data are collected in a consistent, standardised, and reproducible way, for both the control and impact reaches, and before and after remediation:

- Sampling should be carried out when the target fish species are migrating and likely to have reached the site (based on the distance inland).
- Sampling upstream and downstream of the structure should be carried out on the same day and the before and after surveys are carried out at the same time of year (i.e., within the recommended December to March timeframe).
- The same sites are used for each repeat survey.
- Sampling is carried out under similar low flow conditions.

To maximise detection of the species present, it is recommended that for each of the before, after, control and impact samples:

- Utilise the 6 × 1 L replicate or 6 × passive sampler method.
- Samples should be taken at the thalweg (deepest part of the stream) of the stream or as close to the thalweg as possible.
- Samples upstream of the structure should be taken where the stream is unimpacted by the structure itself, i.e., upstream of any impoundment of the stream.

Fish migrations are highly variable and can occur in pulses or triggered by specific environmental cues. Consequently, migration past the structure will also be highly variable and as such, eDNA sampling at one point in time may lead to false conclusions being drawn from once off or short-term sampling. To increase the likelihood of drawing valid conclusions from eDNA monitoring of fish passage remediation success, we recommend the following:

- One pre-remediation sampling at each site.
- Annual sampling at each site for three years post-remediation, with samples taken at the same location, flow conditions and calendar month as pre-remediation sampling.

Indicator species

At each site, the target species for passage may influence when to use eDNA sampling for assessing the effectiveness of barrier remediation over traditional fishing techniques. For example, īnanga is a weak-swimming fish and used as an indicator species for fish passage remediation because if īnanga can pass the obstacle, it is likely that all other species will also be able to navigate the impediment. Īnanga are primarily an annual species with adults migrating downstream to the salt wedge to spawn. Peak spawning typically occurs in autumn and most adults die after spawning. As such, over winter there will be markedly less īnanga DNA present above instream structures that are located upstream of spawning grounds. Consequently, the most īnanga eDNA detected upstream of instream structures during the recommended December to March sampling period is likely to originate from upstream migrating juveniles successfully passing the structure that season. In this regard, where populations are present, īnanga may be an effective indicator species in eDNA monitoring of barrier remediation.

In-situ monitoring

Timing

In-situ monitoring targets fish migrating upstream past the instream structure. As such, monitoring needs to be carried out during the target species' migration period(s). Determining the appropriate timing for monitoring considers not only the migration season, but also the period the target species reach the site. For example, īnanga will reach sites further inland or at higher altitudes later in, or even after, the whitebait season. In the Waikato River, at locations greater than 50 km inland, peak runs of īnanga can occur in December or January. If there is no prior knowledge on when key species reach the site, pilot studies should be carried out, or alternatively, monitoring can be carried out across several months. After determining the appropriate timing to monitor the target species, carry out the monitoring early in the migratory window to ensure the movement of the smallest fish reaching the site are included.

Replicate

All native migratory species have a diel pattern in their migratory movements. For example, all five whitebait species are diurnal with their main migration undertaken during daylight hours (McDowall 2011; Baker and Smith 2015), whereas elvers, bullies and lamprey are nocturnal and mainly move during the night (McDowall 2011). To account for different periodicities across migratory species, a sample replicate of 24 hours is recommended, which is commonly used in Australia for in-situ monitoring (Jones and O'Connor 2017).

Sampling period

Fish migrations are highly variable and can occur in pulses or be triggered by specific environmental cues, as such, migration past the structure will also be highly variable. Figure 8-7 provides an example of the natural variability that occurs in migratory fish movements and the false conclusions that can be drawn from short-term sampling. Monitoring upstream of an instream structure 12 consecutive days before and after remediation shows that banded kōkopu were the only one of four species to exhibit significantly higher passage success after remediation with the structure still an impediment to īnanga. However, if monitoring was only undertaken for five consecutive days before and after the remediation, the data indicate that īnanga and banded kōkopu have significantly higher passage post-remediation. As īnanga are a weak-swimming fish that is commonly used as a representative species for successful remediation of migration barriers, it is important that valid conclusions are drawn from the monitoring data.

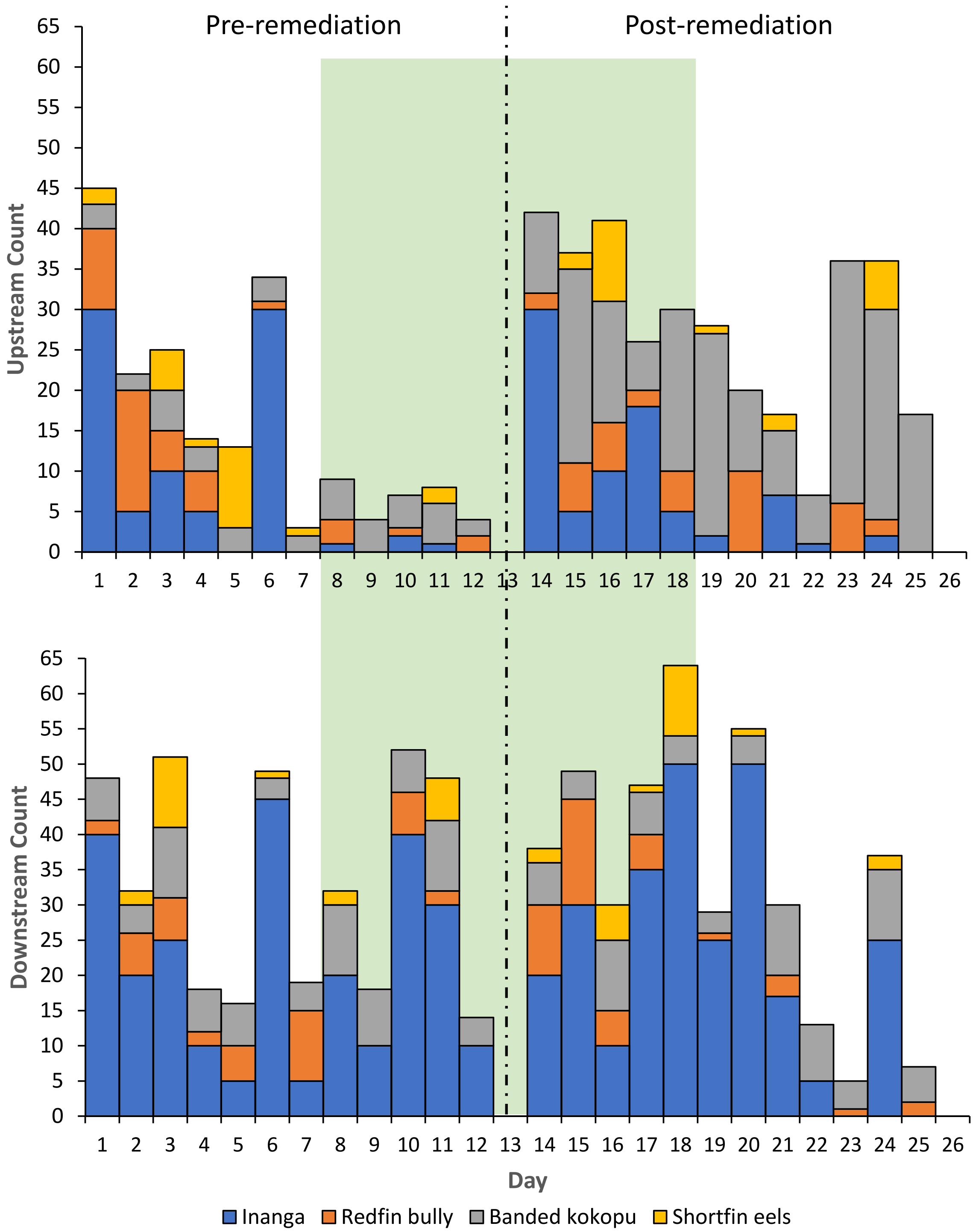


Figure 8-7: Example catch data sampling upstream and downstream of an instream structure 12 days before and 12 days after remediation. The green shaded area represents short term monitoring of five days before and after the remediation. Banded kokopu are the only species of the four shown to have significantly higher passage after 12 days of consecutive monitoring before and after remediation.

To account for the variations in fish movements, a minimum of 12 replicates is recommended with a longer sampling period encouraged. For example, between 14 and 40 replicates have been commonly employed to examine fish passage past vertical-slot fishways (Stuart and Mallen-Cooper 1999; Stuart et al. 2008; Baumgartner et al. 2010). The minimum of 12 replicates needs to be undertaken both *before* and *after* the planned remediation to reduce the likelihood of false conclusions being drawn from the monitoring data (i.e., the sampling design should be balanced). The before trapping should never have a lower number of replicates than the after trapping.

Survey protocol

- Determine the number of days sampling will occur (minimum of 12 days) both before and after remediation of the structure.
- It is recommended to sample the entrance and exit of the instream structure. If multi-barrel culverts are present, then a trap would be set at the upstream inlet of every culvert. To trap downstream of a ford or multi-barrel culvert, setting one double wing fyke net/trap spanning the stream width may be the most practical option.
- Placement of traps, particularly upstream of the structure, should aim to minimise any backwatering effect that could influence water velocities over the structure and bias fish passage results.
- In some situations trapping at the entrance to a structure may not be feasible. Here, trapping the exit location (upstream end of the obstacle) can be undertaken. The recommended 12 replicates should be carried out on consecutive days both before and after remediation (Figure 8-8).
- For the 12 replicates, there is no set protocol for the order in which each end of the structure should be surveyed. The entrance (downstream end) and exit (upstream end) replicates can be randomised across the sampling period, or surveyed alternating between locations. **It is essential that the entrance and exit are not surveyed at the same time as this will confound the results**.
    - A recommended protocol is to trap at the exit then entrance of the structure for four consecutive days at each location, repeating this sequence three times (Figure 8-8). Changing from entrance to exit trapping needs to account for the time taken for fish to pass the instream structure, which will vary according to species, length of structure, modifications present and water flow (i.e., water depths and velocities present). The break in trapping, however, should not allow a pulse of fish to move through/over the structure undetected. Therefore, a 24 h period between entrance and exit trapping is recommended to best account for the differences between sites.
- Trap fish for 24 h periods. Traps should be checked at least once every 24 h period and this timeframe should be utilised as a replicate. In streams with high debris loads, check and clear the trap every 12 h.
    - When clearing the traps: remove all fish and debris accumulated in/on the trap. Process all fish captured to species. For each 24 h sample, record the length of the first 50 fish of each species, counting the rest of the individuals. Once 150 individuals of a species are measured carry out counts for subsequent catches.
- If trapping is carried out across multiple years, then monitoring must occur at the time of the year and within the same month for each repeat survey.

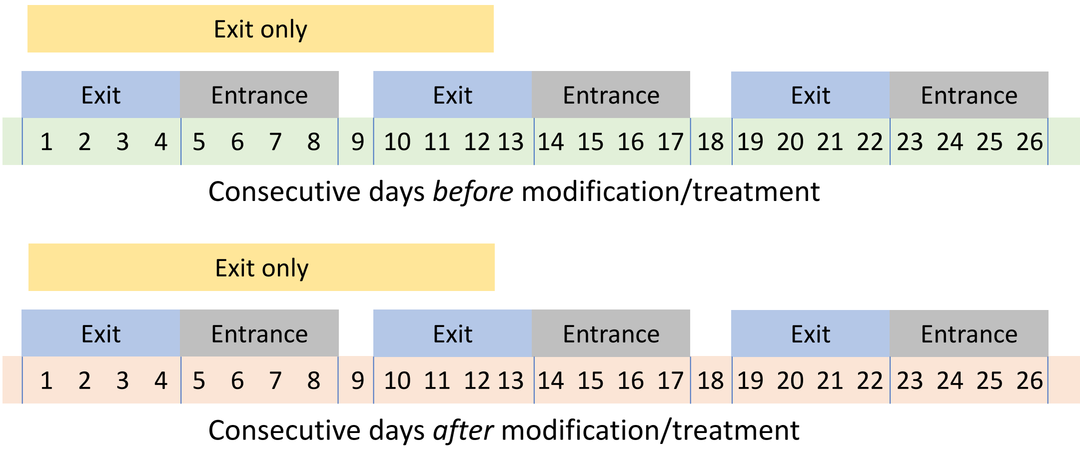


Figure 8-8: Recommended protocol for in-situ monitoring before and after remediation of a fish barrier. The sampling period for trapping at the entrance and exit of a structure and just sampling the exit only is provided. Exit refers to where fish exit the structure (e.g., culvert inlet) and entrance represents where fish enter the structure (e.g., culvert outlet).

Mark-and-recapture

Target species

To ensure the fish pass is effective for all target species, mark-and-recapture trials should utilise the weakest species that requires passage. If passage of swimming fish is desirable, juvenile īnanga are the benchmark species to use if present in the catchment. Common bullies are also a good species to test if present at the site. If passage of climbing fish is the objective, then juvenile redfin bullies are considered the least adept climbing species. If redfin bullies are not present in the catchment, then utilise juveniles of the weakest climbing galaxiid(s) present. Of the four diadromous galaxiids capable of climbing, their ability to surmount instream obstacles in ascending order would be: giant kōkopu, shortjaw kōkopu, banded kōkopu, and kōaro. As obtaining large numbers of identifiable shortjaw and giant kōkopu whitebait is difficult and/or costly, either banded kōkopu or kōaro juveniles are recommended. The location of the instream structure (distance inland) will dictate the target fish species, with juvenile īnanga utilised for structures closer to the coast and banded kōkopu or kōaro juveniles utilised for sites further inland. Multiple target fish can be used and tested if available.

Fish capture and maintenance

It is important to test the life stage of the target species that is expected to be present at the instream obstacle. For example, īnanga reaching many inland culverts will be pigmented, feeding fish (post-whitebait/juvenile) with stronger swimming abilities than fresh-run whitebait captured in the estuary. In this regard, the site of capture for test fish should be representative of the test location.

It is desirable to capture test fish using nets and traps rather than electrofishing. This is to minimise the physiological impact on fish that is likely to influence passage performance.

If possible, setting traps/nets downstream of the test structure to capture fish is recommended. Although the target species is often īnanga or banded kōkopu, capturing a wide range of species present at the site and testing all individuals able to be captured will provide more information about the passability of the structure. If you are collecting fish from a different catchment and transferring them to the structure location, then approval from MPI, DOC and/or Fish and Game is likely required.

To reduce stress and increase performance of the test fish, it is recommended to hold all fish in the stream they are to be tested in. This is because previous trials carried out by NIWA have indicated that fish held in a different water supply to that of the test system, display reduced upstream movement. This loss of motivation could relate to detectable changes in water quality. We recommend holding fish in purpose built live-bins that provide an adequate transfer of fresh aerated stream water (Figure 8-9). Bins should be secured in a pool that provides deep water without excessive water velocities (Figure 8-9). Ensure the lids are cable tied onto the bins otherwise whitebait can push their way out. Test fish should be held for at least 24 hours to habituate and recover from capture and handling prior to colouring in the dye solution. Although experimental releases should be timed with appropriate weather and flow conditions, it is advisable to not hold fish for longer than a week before using in trials.

Figure 8-9: Live-bin deployed to maintain īnanga for fish passage trials. Inset shows close up of live-bin.

Fish marking procedure

Mark test fish by immersion in a solution of Rhodamine B[21] or Bismarck Brown[22]. By colouring fish with two different dyes, it provides two replicates of test fish that can be trialled simultaneously, under the same environmental conditions. In the case study of the Upper Kingston culvert (see Baker et al. (2024a)), where no īnanga could be captured in Kara Stream at the time of carrying out the mark-and-recapture trials, unmarked īnanga could also be released as a third replicate. These fish also act as a control for the marked fish as they have not had the additional stress of staining and are less visible to predators. Unmarked fish should only be used as test fish in situations where fish can be removed from the test reaches and these fish are not naturally occurring in high numbers and, therefore, cannot infiltrate the test reach and confound results.

In a trial evaluating fish passage through a standard single culvert in a wadeable stream, between 100 and 200 fish per replicate would typically be used. However, if only low numbers of test fish are available (e.g., such as banded kōkopu whitebait) then using 30–50 fish per replicate will suffice. At more complex structures, or structures in larger streams (e.g., a weir across a stream), it may be necessary to increase the number of fish used per replicate to increase the probability of capture during the trial.

To stain fish:

- In the shade adjacent to the stream, set up a separate bin containing 50 litres of stream water (to stain up to 1 kg fish) for each dye solution.
- Ensure aquarium salts are added to the solution (sold in pet shops to make salt water) to produce a salinity of c. 15‰. This is vital to buffer the solution, otherwise fish will suffer a high mortality rate. A refractometer is necessary to test the salinity of solution.
- Add 10 g of Rhodamine B (0.2 g/L) or 2.5 g Bismarck Brown (0.05 g/L). Wear gloves when handling the dyes. Refer to the material safety data sheet (MSDS) for each compound to ensure safe practices are adhered to. Rhodamine B colours fish pink, and Bismarck brown colours fish orange (Figure 8-10).
- Aerate the solution well with a portable air supply system. A dive cylinder and adapted regulator or portable 12 volt air compressor unit would be suitable.
- Determine the stream water temperature and add ice as necessary to the dye solutions to maintain the water at ambient stream temperature.
- For fish in Rhodamine B, remove after 2 hours, and for fish in Bismarck Brown, remove after 1.5 hours. Wear gloves while removing fish using a dip net.

[21] http://www.sigmaaldrich.com/catalog/product/sigma/r6626?lang=en&region=NZ&gclid=Cj0KEQiAwPCjBRDZp9LWno3p7rEBEiQAGj3KJgIsyxGXuruPdLVT5O5k7MEP9-rFYmNe--7qRJcTBOIaAkMt8P8HAQ

[22] http://www.sigmaaldrich.com/catalog/product/sigma/15000?lang=en&region=NZ

- After the marking procedure is completed, discard the waste solution according to the protocols outlined in the MSDS[23]. Do not pour it into the stream or on the stream bank. Wear gloves while removing fish and cleaning bins and dip nets.
- Hold coloured fish overnight in live bins to recover before trials.

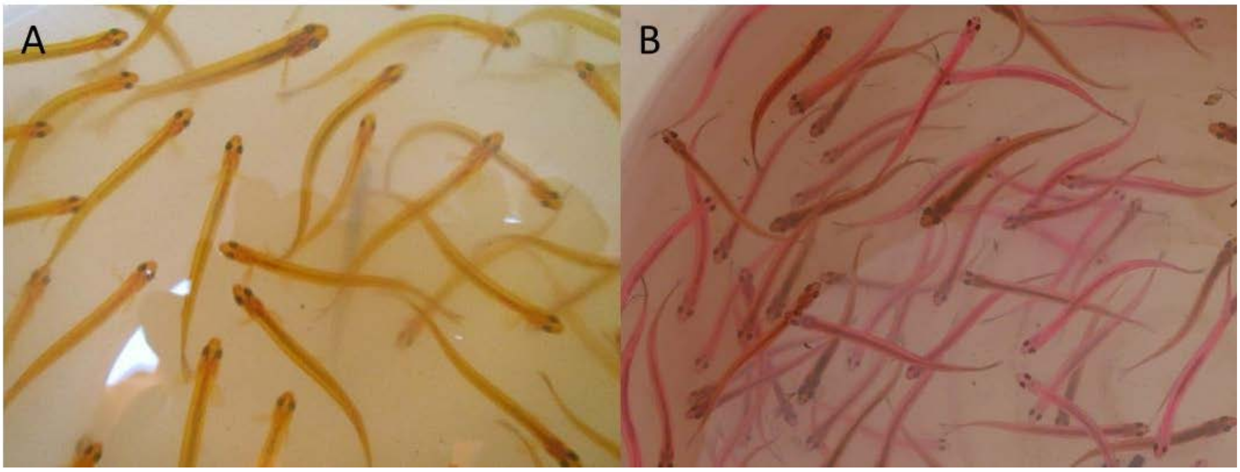


Figure 8-10: Fish coloured orange with Bismarck Brown (A) and pink with Rhodamine B (B).

For each experimental trial, it is advisable to hold 10% of the marked fish in a live-bin as 'control' fish to verify mortality attributable to the colouring procedure.

Stop nets and trap

Install a stop net barricade at the bottom of the test site to prevent fish escaping downstream or stream fish moving upstream. A seine net or whitebait mesh form suitable barriers (Figure 8-11 & Figure 8-12). Alternatively, a double-winged fine mesh fyke net can be used if it spans the full stream width. It is important to dig the bottom of the fyke net mesh or stop net into the substrate and cover with boulders to create a secure barrier. If possible, the top of the fyke net mesh or stop net can be secured to trees on the stream banks (Figure 8-12B), otherwise waratahs or stakes will need to be used (Figure 8-12C). Installing a second net downstream as a back-up is also advisable (Figure 8-12). The barrier should be installed below a pool at the base of the structure to provide fish with a low velocity area to rest before ascent.

If the pool downstream of the structure is too large (i.e. there is a significant perch of the culvert and a deep pool downstream) or is not feasible to barricade the stream for other reasons, then using the barrier net to create a small pool for holding test fish will be necessary (Figure 8-11). Note: it is desirable to create a pool at the base of any remediated fish migration barrier to dissipate energy and prevent erosion.

At the top end of the test site, a whitebait trap and barrier net (or fine mesh fyke net) also needs to be installed (Figure 8-13). Ensure the trap is weighted down to avoid any movement with increases in water flow. For structures with multiple culverts, a separate trap and whitebait mesh should be used at the inlet of each culvert. Once nets and traps are set it is preferable to minimise disturbance of the stream bed within the barricaded area to reduce the likelihood of debris being mobilised and clogging the nets.

[23] msds (fishersci.com); msds (fishersci.com)

Figure 8-11: Sock net used to create a pool downstream of a floating ramp to hold test fish within. Photo credit: Sjaan Bowie.

Figure 8-12: Downstream barricades installed in Kara Stream during the īnanga passage trial. A - C Barrier nets deployed during the rock ramp trial. D, Barrier nets deployed for testing īnanga passage through the culvert independently of the rock ramp.

Figure 8-13: Whitebait trap installed at the culvert inlet in Kara Stream.

Measurements

Flow

It is important to record the flow at the time of the trials. If the study stream does not have a water level recorder installed, a flow gauging can be carried out on each day the trials are being undertaken. Harding et al. (2009) set out basic procedures for stream flow measurement suitable for this purpose.

Water velocity

It is also advisable to measure the average water velocity over each section of the instream structure (e.g., culvert and rock ramp). This will help inform or predict potential problem areas for fish passage, as well as provide some comparative information between sites. The most commonly used method to calculate average water velocity is to time how long a float takes to travel a set distance. A mandarin or orange makes an excellent float as it is easy to see, can withstand knocking into rocks, and it floats almost submerged, so the wind does not influence its movement. It is advisable to measure the average water velocity on each of the trial days.

Trial length

As each instream structure and stream system is different, the appropriate trial length will be determined during the monitoring, but based on results from previous studies, it is recommended that fish are given 24 hours to pass an instream structure. The trap can be inspected after 12 and 24 hours to determine if extending the trial to 36 or 48 hours is warranted.

Sampling protocol

- Initiate trials in the early morning where possible. This may require the barricades to be installed the previous day.
- Prior to releasing the marked fish, electric-fish the test reach to remove any resident fish that could confound trial results. If not marking or tagging pre-fishing is crucial to avoid confounded results. Utilise multi-pass fishing until no fish are captured.
- Release the marked fish at the base of the structure inside the barricade (Figure 8-14).
- Check barrier nets periodically throughout the trial to ensure they remain functional. However, do not walk adjacent to the stream edge to prevent spooking the fish.
- If testing passage over a structure with multiple components, i.e., a culvert and rock ramp, at the conclusion of the trial install a temporary stop net at the base of the culvert to prevent upstream and downstream fish movement between each section of the structure.
- Empty the upstream trap into a bucket or fish bin to hold fish for processing.
- Electric-fish each component of the structure separately, in a downstream direction to collect fish that failed to pass. Use multi-pass fishing until no fish are collected over several passes. Keep fish collected from each section of the structure in a separate bucket.
- Anaesthetise fish in each bucket (successful, unsuccessful and in transit) and record their length and colour. Alternatively, utilise a photarium to view and measure

individual fish. If time allows, record the length of every recaptured fish, otherwise ensure lengths are measured for at least 50 successful and 50 unsuccessful fish from each replicate (e.g., pink, orange and unmarked) in each bucket (i.e., successful, unsuccessful and in transit). This will determine if fish size influenced passage success over the instream structure. Count the remaining fish where lengths are not measured.

Figure 8-14: Releasing marked īnanga below the rock ramp in Kara Stream, at Upper Kingston Road.

8.2.5 Survey timing

BACI fishing survey

At any given site, there is considerable temporal variation in most fish species' abundances. This is largely due to annual variation in the recruitment of diadromous fish species, and the seasonal migration and movement patterns of different fish species. In addition, abiotic and biotic factors can influence the efficacy of a given species capture rates. For these reasons we recommend carrying out fish surveys **between December and April inclusive**, with any repeat monitoring carried out in the same month each survey.

The December to April timing ensures sampling is carried out when fish are most active, however, it is important to note that species such as smelt and īnanga undertake downstream spawning migrations to the lower river/estuary during autumn and early winter. The timing of these downstream spawning migrations are contingent upon latitude, but peak movements generally occur between March and May. In this regard, if smelt and īnanga are target species, then monitoring should be carried out between December to February inclusive.

Mark-and-recapture surveys

A critical aspect of mark-and-recapture trials is timing, with two key factors to consider: season and flow. As migratory whitebait are usually used as target species for mark-and-recapture trials, we recommend carrying out trials **between September and March inclusive.** This window represents the peak upstream migration season and the timing that whitebait recruit into streams across summer**.** We do not recommend testing īnanga whitebait later than March as the downstream spawning migration undertaken by adult fish can be initiated and confound trial results.

Site location will be important when considering trial timing. For sites close to the coast where whitebait, and particularly īnanga, reach the site from August/September, then mark-and-recapture trials utilising īnanga whitebait are best carried out in September or October to reflect the performance abilities of fresh sea run fish. If whitebait are not arriving at the site until fully pigmented feeding fish, then mark-and-recapture surveys should reflect the timing fish usually arrive at the site. Capturing the test fish from close to the instream structure will ensure the appropriate life stage is tested. Repeat monitoring should either be carried out in the same month, or within four weeks of the control/baseline survey.

It is crucial to carry out the trials during base flow in the study stream, under a high pressure front that will limit rainfall and subsequent rises in stream discharge over the trial period. This is not only because the barricades and trap can get washed out, but also because instream structures are generally harder for fish to pass at low flows. In this regard, trials carried out at base flow won't overestimate the proportion of fish able to successfully pass the structure.

8.2.6 Defining success

The performance of any fish pass will vary with the type of pass and target species, as well as specific site conditions. As highlighted with the case study at Bankwood Stream (see Baker et al. (2024a)), fish passage performance can vary according to the size and condition of the fish as well as with environmental variables such as flow. The relationship between passage performance and flow will likely change throughout the migration season and between years, and this needs to be considered when interpreting passage success. Although the efficiency of a fish pass is a quantitative measure of its performance, it needs to be considered in the context of the efficiency required to maintain upstream communities. In general, for any site and species, the two main factors influencing the required efficacy of passage past the structure will be the carrying capacity of the upstream habitats and the number of recruits reaching the base of the structure. In Bankwood Stream, approximately 30% passage efficiency of īnanga past the culvert is maintaining species such as smelt and īnanga in the upstream habitats. However, because of an additional migration barrier to non-climbing fish species, only around 160 m of linear stream is currently accessible to swimming fish species, meaning that upstream habitat is limited.

The results should also be considered in a catchment context. The cumulative effect of individual fish passes or structures can have a multiplicative impact on the proportion of successful fish recruits reaching upstream habitats. This is illustrated in Figure 8-15 for several hypothetical examples of multiple structures with passage efficiencies of 10%–90%. For example, if upstream migrants are required to pass a series of five culverts, where passage efficacy at each culvert is 50%, then only 3.1% of fish will successfully reach upriver habitats. Consequently, passage efficiency at each individual structure may need to be higher to account for the cumulative effects of multiple structures on the fish community composition.

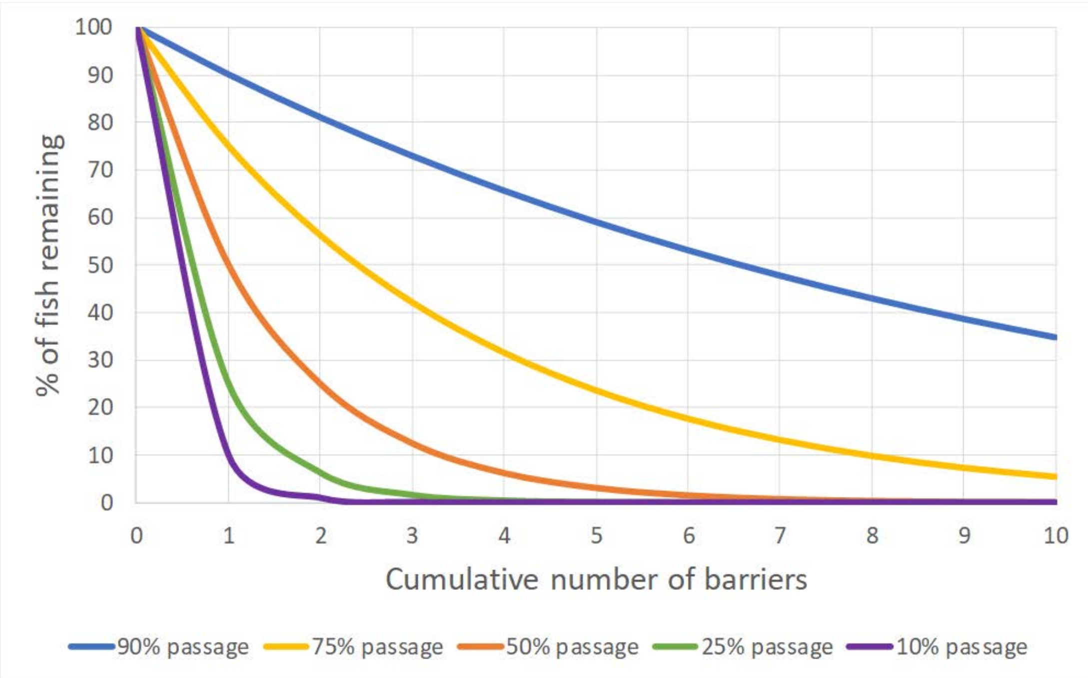


Figure 8-15: Illustration of hypothetical cumulative effects of multiple instream barriers with varying passage rates.

Too few fish pass solutions have been monitored at present to provide guidelines on the required passage efficiency necessary to maintain upstream fish communities relative to distance inland and carrying capacity of different sized catchments. Consequently, moving forward it will be important to carry out a robust monitoring programme (i.e., pairing BACI surveys with mark-and-recapture trials) for new or remediated instream structures to improve determination of the efficiency required to define successful passage across a range of structures and situations. The appropriate threshold will likely vary depending on life stage, stream habitat availability, location in the catchment and the species present.

As a rule of thumb, for any waterway, a remediated or new instream structure should pass all fish species and life stages present under all migration flows or flows upon which movements between critical habitats are carried out.

8.3 Physical and hydraulic monitoring

Means objectives for fish passage projects may include physical and/or hydraulic objectives alongside biological objectives. Monitoring of physical/hydraulic parameters is often more straightforward than evaluating biological performance measures and is an important way to identify any maintenance requirements. However, **it is important to recognise that achieving physical/hydraulic performance standards does not guarantee achievement of associated biological objectives and performance standards**. As such, physical/hydraulic monitoring should be deployed alongside biological monitoring techniques rather than being considered as an alternative to biological monitoring.

Physical/hydraulic performance measures are typically used for two main purposes:

1. checking a newly constructed fish pass or instream structure against design specifications, and
2. as part of an ongoing risk-based surveillance/maintenance monitoring programme.

8.3.1 Evaluation of newly constructed fish passes or instream structure

After a new fish pass or instream structure has been completed, a range of physical/hydraulic measurements should be taken to ensure that the constructed fish pass or instream structure is consistent with the design specifications and tolerances established during the detailed design phase and/or guidelines for design (see Section 4). The purpose of the commissioning assessment is to ensure that the fish pass physical/hydraulic parameters accurately reflect the design/guidelines specifications. For all new structures it is also a requirement of the NES-F that the required information is submitted to regional councils.

Jones and O'Connor (2017) identified and defined a range of hydraulic parameters relevant to evaluating fish pass designs as summarised in Table 8-5. Measurements should be taken at all pools and baffles to check for consistency across the fish pass. In most cases, a tolerance of ≤5% departure from the design/guidelines will be acceptable, but for technical fishways (e.g., vertical slot fishways) deviation in key parameters such as the pool volume, slot width, slope and head loss should be constrained to within ≤2%. In addition to measurements of physical/hydraulic conditions within the new fishway, it may also be important to take measurements of key physical/hydraulic parameters within or across associated infrastructure (e.g., the culvert that is being remediated) that are also important for ensuring fish are able to pass the structure. Key parameters would include water depth and water velocities inside the culvert, for example.

It is recommended that commissioning assessments be carried out across the specified operating range of the structure to ensure that performance remains within design specifications throughout the operating range. These measurements will typically require use of tape measures, a water velocity meter, and a laser level and measuring rod. Once it has been established that the fish pass meets the physical/hydraulic design specifications, biological evaluation can commence.

Table 8-5: Example of physical/hydraulic performance measures relevant to assessing fish pass design. Modified from Jones and O'Connor (2017).

Parameter	Definition
Target water depth	Water depth of each pool and at the slot
Minimum pool volume	Volume of the pool based on minimum target depths (L × W × H)
Minimum slot width	Width of the slot at all baffles
Maximum water velocity at vena contracta	Maximum water velocity at the vena contracta (i.e., jet of water at the slot)
Slope	Slope of the fish pass between the entrance and exit
Head loss	Difference in water height between pools
Minimum head loss at fishway entrance	Difference between river height at entrance and first pool
Maximum water velocity at fishway exit	Maximum water velocity at the exit
Entrance and exit flow vectors	Angle/direction of flow at the entrance and exit
Entrance/exit location	Location of the entrance/exit at the upstream migration limit

Parameter	Definition
Water velocity within culvert	Average and maximum water velocities within the culvert
Water depth within the culvert	Average and minimum water depth within the culvert
Culvert substrate	Visual assessment of percent coverage of different substrate types within the culvert

8.3.2 Risk-based physical/hydraulic surveillance monitoring

An advantage of physical/hydraulic monitoring over biological monitoring is that physical/hydraulic parameters are often (relatively) easily and quickly measured. As such, they can play a valuable role as part of a risk-based surveillance monitoring framework. In this context, physical/hydraulic measurements are taken on an ongoing basis to determine whether the structure or fish pass remains within the design specifications or guidelines for the site, or whether change is occurring over time. Where changes in the physical/hydraulic performance measures are identified, this can be used as an indicator of an increased risk that fish passage is being impeded and trigger more detailed investigations and/or biological monitoring, or management interventions.

The Fish Passage Action Plan Template (Ministry for the Environment 2022) suggests all instream structures should be checked annually and/or after significant natural events to ensure that they continue to meet the required fish passage objectives of the NPS-FM. It is unrealistic and, in most cases, unnecessary to require that biological monitoring be undertaken annually on an ongoing basis or following every significant natural event. However, physical/hydraulic surveillance monitoring is often practicable and could be carried out alongside/as part of standard infrastructure maintenance checks.

The NES-F requires certain information to be provided to the regional council within 20 days of installation (see Section 2.4.1). The Fish Passage Assessment Tool (FPAT; https://fishpassage.niwa.co.nz/) can be used as one tool for tracking basic information about the physical characteristics of structures that is relevant to fish passage risk. However, the FPAT was not designed to capture detailed information on remediation interventions like the physical/hydraulic performance measures set out in Table 8-5. As such, more targeted physical/hydraulic monitoring aligned with design or guidelines specifications may be more suited for ongoing surveillance monitoring of fish passage interventions. Where the selected physical/hydraulic performance measures are found to be consistent between assessments and aligned with the design specifications, an assumption can be made that overall performance remains consistent. Where changes are identified over time, particularly where they exceed physical/hydraulic performance standards (i.e., they fall outside of the design specifications), more detailed assessment should be triggered, including biological assessments as outlined in Sections 8.2 to 8.2.6.

8.4 Cultural monitoring

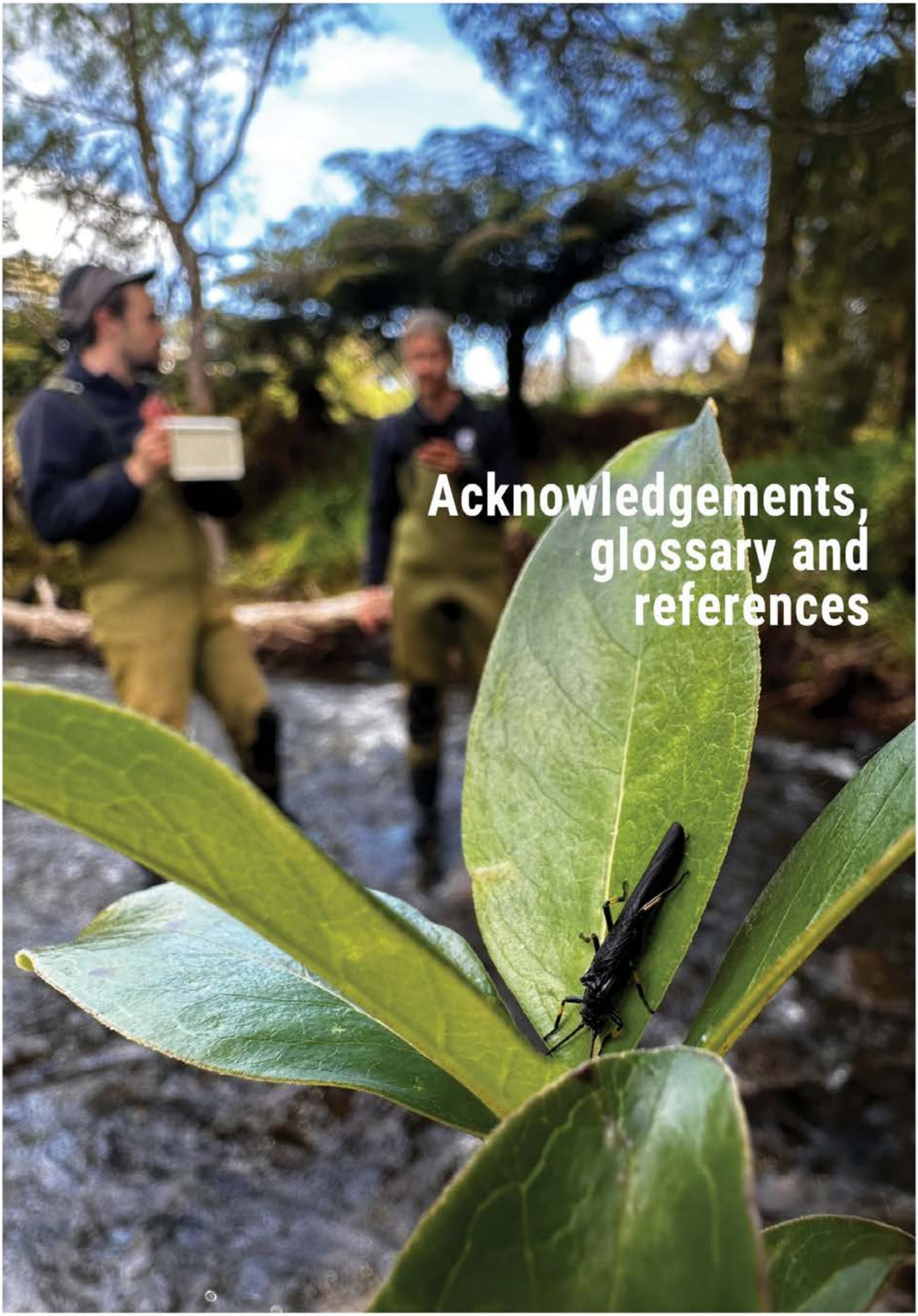
Each hapū and iwi have their own mātauranga that is specific and relative to their environmental contexts, experiences, observations and understandings of their interactions and patterns of use.

Hapū/iwi-driven cultural monitoring programmes can add value to freshwater monitoring initiatives, providing a more holistic and integrated assessment of ecosystem health and wellbeing. The methods and processes used by iwi/hapū to describe their values, assess the baseline state of these values and the pressures impacting on them, as well as identifying appropriate tools and approaches to monitor changes in state at temporal and spatial scales of relevance to them, involves its own processes and methodologies. 'Monitoring' is generally one component within a strategic workplan that may be implemented by iwi and hapū to support their co-management of environmental resources.

Many iwi/hapū have produced guidance around their expectations for their involvement in resource consent processes and environmental monitoring. For example, Te Korowai o Ngāruahine Trust state that "Cultural monitoring is undertaken by Hapū to protect and manage sites of significance at a cost to the regulatory authority or consent applicant. Te Korowai o Ngāruahine Trust or Hapū may request the engagement of cultural monitors under various circumstances" (Te Korowai o Ngāruahine Trust 2021).

Ngāruahine describe the purpose and outcomes sought by cultural health monitoring as follows: "Cultural Health Monitoring shall be undertaken to identify and articulate values and perspectives of environmental change, and to assess the mauri of freshwater, soils, coastal water quality, mahinga kai and mātaitai. Monitoring is undertaken by Hapū to provide a tangata whenua perspective on changes to the Taiao based on traditional oral baselines of mauri. Using mātauranga Māori links the health of the environment to the health of the people and provides important information which can be used in parallel to western science monitoring. Indicators used in cultural health monitoring will be determined by Hapū with the support of Te Korowai o Ngāruahine Trust" (Te Korowai o Ngāruahine Trust 2021).

Iwi/hapū may also identify performance indicators relevant to their objectives and aspirations. For example, the Ngāi Tahu Freshwater Policy sets out relevant performance indicators including mauri, water quantity, water quality, and mahinga kai indicators (Te Rūnanga o Ngāi Tahu 2015). Ngāi Tahu Papatipu Rūnanga have implemented various cultural environmental monitoring initiatives, including the Cultural Health Index for Streams and Waterways (Tipa and Teirney 2006), the State of the Takiwā – Te Ahuatanga o te Ihutai (Lang et al. 2012), Cultural Flow Preference Studies (Tipa & Associates 2016), and the Murihiku Cultural Water Classification System (Kitson et al. 2018). Such monitoring frameworks and indicators may offer a complementary means of evaluating the outcomes of fish passage remediation initiatives.



Acknowledgements, glossary and references

9 Acknowledgements

We appreciate the thoughtful and constructive feedback from stakeholders on the original version of the guidelines that has shaped our thinking during this update. We also value the time that various people have put into providing feedback on early versions of this revision. We would particularly like to acknowledge Tim Marsden (Australasian Fish Passage Services) and Ivor Stuart (Charles Sturt University) who have been generous in sharing their knowledge and practical experience of implementing evidence-based fish passage remediation.

We are grateful to all those who answered our call for new information and data on the performance of different fish passage solutions in New Zealand. Ensuring that we have the latest information is invaluable in being able to provide users with up-to-date evidence-based guidelines. We would also like to acknowledge the contributions of Kristina Mead and Sarah Bevin (Tonkin + Taylor Ltd) who provided valuable input on the updates to the legislation section and Ngahuia Herangi, Erica Williams, Tessa Thomson, and Tekiteora Rolleston-Gabel (NIWA) who supported creation of the te ao Māori case studies now incorporated in the guidelines.

We value the constructive and thoughtful review comments provided by Justin O’Connor (Arthur Rylah Institute) on improving the remediation section and evidence syntheses and the time taken by the NIWA reviewer Mike Hickford to thoroughly review the final draft of a large document. We also recognise the important contribution of Jo Downey (NIWA) who had the unenviable job of sorting out all the formatting.

Finally, we appreciate all those who have kindly contributed the photos that we have included in the guidelines and the great final production work from the NIWA Comms team.

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10 Glossary of abbreviations and terms

10.1 Technical glossary

Amphidromous	Amphidromous fish hatch in freshwater/estuaries, then drift into the ocean as larvae before migrating back into freshwater to grow into adults and spawn, e.g., banded kōkopu
Anadromous	Anadromous fish hatch in freshwater, migrate to the ocean as juveniles where they grow into adults before migrating back into freshwater to spawn, e.g., lamprey
Ancillary structure	Ancillary structures include additional features such as headwalls, wingwalls and aprons that may be required to complete the construction of a primary structures such as a culvert or weir
Apron	A hardened surface (usually concrete) placed at the inlet and/or outlet of a structure to protect the structure from erosion
Attraction flows	The flow of water required to direct moving fish towards a fish pass or bypass channel
Backwatering	The effect of backing up water in its course by an obstruction
Baffles	A device used to modify and restrain the flow of water
Bank-full discharge	The river flow that just fills the stream channel without overtopping the banks. This is generally considered the dominant channel-forming flow
Bank-full elevation	The water level at bank-full discharge
Bank-full width	The wetted width at the bank-full discharge
Bed	Refer to RMA Part 1 Interpretation and application
Broad-crested weir	A weir with a crest of significant thickness measured in the direction of flow
Exclusion barrier	An instream structure built with the explicit intent of restricting or preventing the movement of aquatic organisms
Bypass structure	A structure used to facilitate fish movements around instream obstructions. They are often known as fish passes or fishways
Catadromous	Catadromous fish hatch in saltwater, then migrate into freshwater as juveniles where they grow into adults before migrating back into the ocean to spawn, e.g., longfin eel
Critical shear stress	The minimum amount of shear stress exerted by stream flow that is required to initiate movement of substrate particles
Culvert	A connection between two water bodies or parts of a waterbody, typically a pre-formed concrete tube located below roads or other constructions

Dam	Any structure designed to confine, direct, or control water, whether permanent or temporary; and includes weirs
Denil fishway	A type of technical fishway consisting of a linear channel in which baffles are arranged at regular and relatively short intervals, angled against the direction of flow
Diadromous	A category describing fish that spend part of their lives in freshwater and part in saltwater. Anadromous, amphidromous, and catadromous are all sub-categories of diadromous
Diversion structure	Any structure designed to divert or abstract natural water from its natural channel or bed whether permanent or temporary
Fish facility	Any structure or device, including any fish pass or fish screen inserted in or by any water course or lake, to stop, permit, or control the passage of fish through, around, or past any dam or other structure impeding the natural movement of fish upstream or downstream
Fish pass	Any structure providing passage through or over any barrier to fishes' passage
Fish passage	The movement of fish and other aquatic organisms between all habitats necessary to complete their life cycle
Fish passage design flow	The range of flows over which fish passage is required
Fish screen	Any device whether moving or stationary designed to impede or stop the passage of fish
Fishway	See Fish Pass
Ford	A shallow place in a river or a stream allowing one to walk or drive across
Head drop	The difference between water levels upstream and downstream of a structure
Hypoxia	Oxygen deficiency in the environment
Impede	Delay or prevent by obstructing them; hinder
Nappe flow	The term nappe refers to the sheet of water flowing over a weir crest. Nappe flow occurs when the sheet of water is not in contact with the weir structure (i.e., there is an air gap between the underside of the nappe and the downstream weir face)
Nature-like fishway	A bypass structure that mimics natural stream characteristics in a channel that bypasses a barrier
Open channel design	A design process using the principles of open channel hydraulics. Open channel hydraulics is a branch of fluid mechanics dealing with the conveyance of water through conduits with a free surface (i.e., the surface of the water is in contact with the air and not under pressure)

Overshot weir	A weir where water flows over the top of the weir
Passage	The action or process of moving through or past somewhere on the way from one place to another
Peak design flow	The highest flow that a structure is designed to convey
Pool and weir fishway	A type of fish pass consisting of a series of small dams and pools of regular length to facilitate the movement of fish around or over an obstruction
Remedial works	Any structures, channel modifications, or water flow provided to offset the effect of a dam or diversion structure
Rheotaxis	An innate behaviour in fish that leads them to orientate themselves into the flow
Rock ramp fishway	A type of fish pass consisting of rock ridges and pools that mimics natural stream conditions to facilitate movements of aquatic organisms around or over an obstruction
Shear stress	A measure of the force of friction from a fluid acting on a body in the path of that fluid
Subcritical flow	Flow with a velocity lower than the wave velocity (i.e., surface ripples progress upstream as well as downstream). Downstream influences can cause upstream effects. Flow is typically deep and slow
Supercritical flow	Flow with a velocity higher than the wave velocity (i.e., surface ripples do not progress upstream). Downstream influences do not cause upstream effects. Flow is typically fast and shallow
Technical fishway	A category of fish pass generally characterised by a relatively formal structure. Typically dependent on quite strict hydraulic design criteria to provide conditions suitable for passage of the target fish species. Examples include vertical slot and Denil fishways
Undershot weir	A weir where water flows underneath a weir gate. These are sometimes referred to as sluice gates
Vertical slot fishway	A type of fish pass consisting of a series of pools separated by walls with a narrow vertical gap allowing fish to pass between pools
Weir	A barrier across the cross-sectional width of a river that alters the flow characteristics of the water and usually results in a change in the height of the river level
Weir crest	The top edge of a weir that water overflows
Weir face	The downstream sloping face of a weir
Wetted margin	A shallow, low velocity area along the edges of the water
Wetted width	The width of the river channel at the water surface
Wingwall	A wall on a structure that ties the structure to the river bank

10.2 Te reo Māori glossary

Aua	Yellow-eyed mullet
Hao	To catch in a net, gather together, net, harvest; shortfin eel
Hapū	Kinship group, clan, tribe, sub-tribe, extended family – often refers to a subtribal/extended family kinship group, that consists of extended family who descend from a common ancestor.
Iwi	Extended kinship group, tribe, nation, people, nationality – often refers to a large group of people descended from a common ancestor and associated with a district territory .
Kaitiakitanga	The intergenerational exercise of customary custodianship, in a manner that incorporates spiritual matters, by those who hold mana whenua/moana status for a particular area or resource.
Kākahi (kāeo/waikākahi/torewai)	Freshwater mussels
Kaupapa	Topic, policy, matter for discussion, plan, purpose, scheme, proposal, agenda, subject, programme, theme, issue, initiative.
Kaitiaki	Guardian, steward, custodian.
Kōura (kēwai/waikōura)	Freshwater crayfish
Mahi	Work, task, practise.
Mahinga kai (mahika kai)	Refers to the species that have traditionally been used as food, tools, medicine, or other resources, including the act of harvesting/practice/use of those resources and the places they are gathered.
Maramataka	Māori lunar calendar - a planting and fishing monthly almanac
Mata	Whitebait
Mātauranga	Is a holistic perspective encompassing all aspects of Māori knowledge and seeks to understand the relationships between all component parts and their interconnections to gain an understanding of the whole system. It is based on its own principles, frameworks, classification systems, explanations, and terminology. It captures both traditional knowledge as well as new knowledge being created every day in Māori communities. Mātauranga Māori is a dynamic and evolving knowledge system, has both qualitative and quantitative aspects, and includes the processes for acquiring, managing, applying, and transferring that body of knowledge
Morihana	Carp, goldfish.
Paraki	Smelt
Pātiki	Flounder
Piharau (kanakana)	Lamprey
Rohe	Boundary, district, region, territory, area, border (of land, of water, of ocean).
Taonga	Treasures of cultural and historical significance to Māori, e.g., can include species of indigenous flora and fauna.
Te reo Māori	Māori language.

Te ao Māori	Māori worldview.
Tikanga	Māori customary law, values, and practices. Encompasses the correct procedure, custom, lore, method, and practice. The customary system of Māori values and practices or set of protocols that have developed over time and are deeply embedded in the social context.
Tuna	Longfin and shortfin eels.
Wāhi tapu	Sacred site – typically a place subject to long-term ritual restrictions on access or use.
Wāhi tupuna	Usually refers to a place of ancestral significance and associated cultural and traditional values.
Whakapapa	Genealogy
Whānau	An extended family, family group, or a familiar term of address to several people.

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Appendix

Appendix A Ecological considerations for instream structure design

Barriers to fish migration at road crossings and other instream structures can adversely affect fish populations, reducing fish numbers and altering fish species diversity within catchments by obstructing movement and migration to critical habitats. This section explains the importance of freely accessible and connected freshwater habitats for sustaining our valued freshwater fish communities, and highlights some of the key characteristics of instream structures that can impede fish and other freshwater species' movements.

Linking habitats and fish movement

Why do fish and other aquatic organisms need to move?

Many of our native fish species must travel between marine and freshwater environments to complete their life cycle, i.e., they are diadromous. The majority of the most widespread native fish species that occur in New Zealand's waterways have larvae that rear in the sea and then migrate back into freshwater as juveniles. Their adult populations are, therefore, dependent on the success of the annual upstream migrations of juveniles. Some of the main life cycles used by New Zealand fish species are explained below.

Īnanga are the most common of the five whitebait species and are found throughout New Zealand. Īnanga have a catadromous life cycle because their adults migrate from rivers and streams to estuaries to spawn (Figure A-1). The eggs are laid during high spring tides in the intertidal vegetation and develop out of water. After hatching, larvae migrate to the sea to feed and grow. Īnanga migrate back into freshwater as juveniles in search of habitat suitable for growing into adults. This is when people catch them as whitebait. Both longfin and shortfin eels are also catadromous, but their adults migrate all the way to the ocean to spawn.

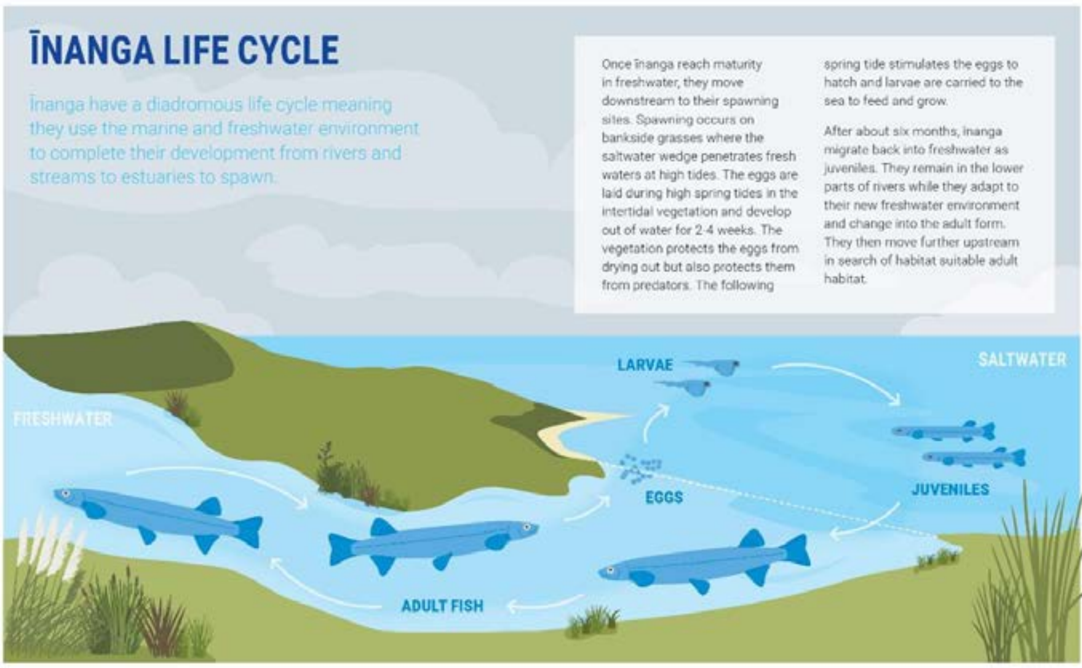


Figure A-1: Life cycle of īnanga. Adults migrate down to estuaries to spawn. Upon hatching, larvae move out to sea and rear into juveniles before returning to freshwater for growth to adulthood.

The other four galaxiid fish species that make up the whitebait catch in New Zealand, banded kōkopu, giant kōkopu, shortjaw kōkopu and kōaro, all have an amphidromous life cycle (Figure A-2). This means the adults do not migrate to marine waters to breed and, instead, spawning occurs in freshwater rivers and streams. The eggs are laid in riparian vegetation on the banks during flood flows and, like īnanga, subsequently develop out of water. Upon re-inundation, the larvae hatch and migrate out to sea to feed and rear. They then migrate back into freshwater as juveniles in search of habitat for growth to adulthood. This type of life cycle is also seen in many of our bully species and torrentfish (*Cheimarrichthys fosteri*).

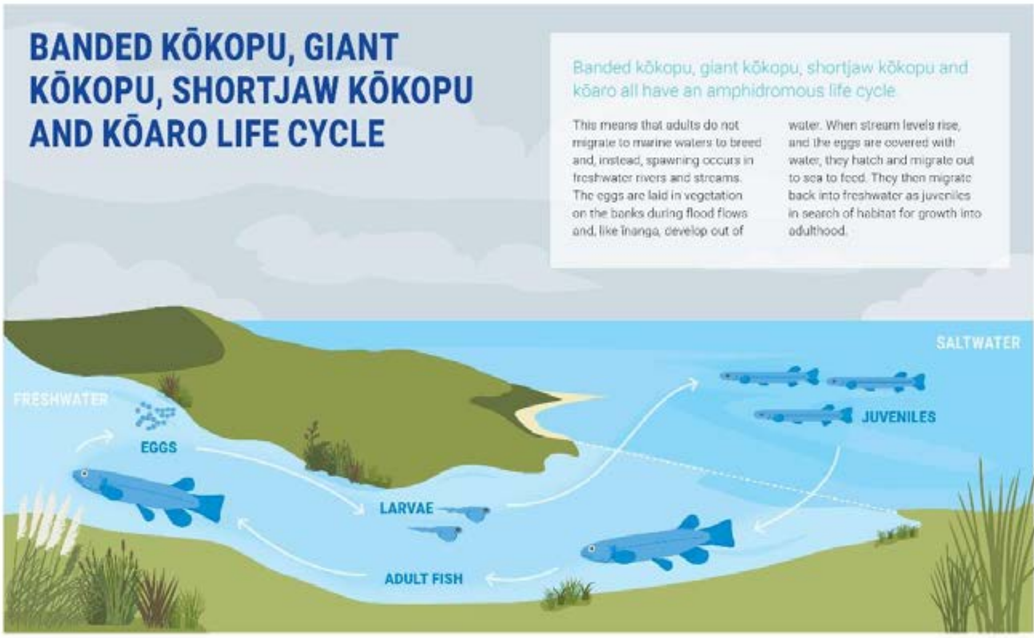


Figure A-2: Life cycle of banded kōkopu, giant kōkopu, shortjaw kōkopu and kōaro. Adults spawn in rivers and streams. Larvae migrate to the sea upon hatching, where they feed and grow into juveniles before returning to freshwater for growth to adulthood.

Some of the species with an amphidromous life cycle also have populations in New Zealand that complete their entire life cycle in freshwater (e.g., banded kōkopu and common bully). These populations are known as lacustrine (i.e., lake-based) or landlocked (i.e., they can't or don't access the sea). After the larvae hatch in tributary streams, rather than moving out to sea, they move to downstream lakes to rear. As juveniles they move out of the lakes again and into nearby streams where they then grow into adults. Despite not undertaking a migration to sea and back, these fish still require connectivity between larval rearing habitats in the lakes and adult rearing and spawning areas in streams.

The lamprey has an anadromous life cycle. This means that their larvae rear in freshwater and migrate to the ocean as juveniles. They feed and grow to adulthood in the ocean and then migrate back to freshwater to spawn and die. Naturally, most salmon and trout species also have an anadromous life cycle. However, in New Zealand most salmonids are non-diadromous and complete their entire life cycle in freshwater. Typically, salmonids spawn in streams, where the larvae hatch and rear. As juveniles they migrate downstream to adult rearing habitats, either in larger rivers or lake systems. In New Zealand, anadromous populations of brown trout and chinook salmon exist in some river systems, but the other salmonid species are not known to have sea-run populations here.

Several native fish species are resident in freshwater, e.g., non-migratory galaxiids, some bullies and mudfish. They complete their whole life cycle in freshwater streams and rivers. These fish still need to move within waterways to varying degrees to access different habitats, e.g., downstream dispersal of larvae, so effective fish passage management remains important.

This diversity of life-history strategies means it is important to understand what fish are present in any location before devising appropriate strategies for providing effective fish passage. This must account for differences in species, life stage, direction, and timing of movements. For example, some species, e.g., giant bullies, rarely move far inland from the coast, but species such as longfin eel and kōaro regularly penetrate a long way inland. Information on what species are present at a site may be available from the New Zealand Freshwater Fish Database (NZFFD)[24]. However, consideration must be given to the timing and methods used for the surveys included in the NZFFD, and the best way to find out what fish are present (or should be present in the absence of barriers) is to undertake a fish survey. There are also many locations where no data are available and, in this case, modelled information on expected fish occurrence may be of use.

The need for free movement is not only limited to freshwater fish, but also many of our aquatic invertebrates. The loss of physical habitat caused by installation of instream structures impacts on the abundance of aquatic invertebrates. There is also evidence demonstrating impacts on adult flight paths, with the presence of culverts being associated with significant reductions in some species upstream of culverts (Blakely et al. 2006). Furthermore, a recent study has shown that recolonisation of freshwater mussels was enhanced following the installation of a fishway at a weir (Benson et al. 2018). Mussels have an obligate larval stage that parasitises fish hosts. If fish movements are limited by a barrier, the dispersal of the mussels at that larval stage is also limited.

Timing of fish movements

The timing of fish migrations vary within and between species. However, the main migrations are typically associated with key stages in fishes' life histories, e.g., spawning, hatching, and rearing. Many of New Zealand's native fish species undertake their main upstream migrations as relatively weak-swimming, small-bodied juveniles. This contrasts with many of our sports fish (e.g., trout and salmon), which undertake their main upstream migration as large, strong-swimming adults. The migration times of some of the main migratory freshwater fish species found in New Zealand are summarised in Table A-1. It is important to consider upstream and downstream movements for resident and migratory species. This information can be used to inform expectations on what species and life stages of fish you might expect to be migrating at any given time and, therefore, inform design criteria for instream structures and timing of installations. However, it should be noted that there are regional variations in the timing of migrations, and it is important to confirm this information locally.

Fish swimming behaviours

The ability of fish to migrate upstream is influenced by several factors including swimming ability, behaviour, and environmental factors, such as water temperature. There are four main modes of movement utilised by fish (Table A-2). Swimming is the primary mode of movement but some species have developed additional modes to help them overcome natural obstructions such as waterfalls and rapids. In New Zealand, several of our native fish species, e.g., eel, banded kōkopu and kōaro, are excellent climbers as juveniles. This allows them to negotiate some obstacles, such as waterfalls, if a continuous wetted margin is available for them to climb and access habitats far inland and at relatively high elevations.

[24] https://www.niwa.co.nz/our-services/online-services/freshwater-fish-database

Table A-1: Freshwater fish migration calendar for key New Zealand migratory fish species.

Showing migration range (light blue □) and peak periods (dark blue ■), migration direction and life stage at the time of migration. ○ Not threatened; ● At risk-Declining; + Threatened-Nationally vulnerable; Δ Introduced sports fish. Life stages: L = larval, J = juvenile, A = adult. * indicates the life stages that are present only within the lower reaches of rivers and streams. Modified from Smith (2014).

Functional group	Species	Conservation status	Direction	Life stage	Summer			Autumn			Winter			Spring		
					D	J	F	M	A	M	J	J	A	S	O	N
Bullies (fast flow) & torrentfish	Bluegill bully	●	↑	J	□	□	□	□								□
			↓	L	□	□	□	□						□	□	□
	Redfin bully	●	↑	J	□	□	□	□								□
			↓	L	□	□	□							□	□	□
	Torrentfish	●	↑	J					□	■	■	■	■	□	□	□
			↓	L	□	□	■	■	■	■	□					
Bullies (slow flow)	Common bully	○	↑	J	□	□	□	□								
			↓	L	□	□	□	□						□	■	■
	Giant bully	○	↑	J	□	□	□	□								
			↓	L*	□	□	□	□						□	□	□
Eels	Longfin eel	●	↑	L*								□	■	■	■	□
			↑	J	■	■	■	■	□							□
			↓	A				□	□	□	□					
	Shortfin eel	○	↑	L*	□								□	■	■	■
			↑	J	■	■	■	■	□							□
			↓	A			□	□	□	□						
Īnanga & smelt	Īnanga	●	↑	J						□	□	□	■	■	■	■
			↓	A	□	□	■	■	■	■	□	□				
			↓	L*	□	□	□	■	■	■	■	■	□		□	□
	Common smelt	○	↑	J									□	■	■	□
			↓	L		□	□	□	■	■	■	□	□			
Lamprey	Lamprey	+	↑	A					□	□	□	□	■	□	□	□
			↓	J	□	□	□	□	□	□	□	□	□	□	□	□
Large galaxiids	Banded kōkopu	○	↑	J	□								□	■	■	□
			↓	L				□	□	□	■	■	□	□		
	Giant kōkopu	●	↑	J	□										□	■
			↓	L						□	■	■	□	□		
	Kōaro	●	↑	J										■	■	□
			↓	L						■	■	□	□	□		
	Shortjaw kōkopu	+	↑	J										□	□	□
			↓	L						□	■	■	□			
Salmonid sports fish	Atlantic salmon	Δ	↑	A	□	□	□	□	□	□						
			↓	J						□	□	□	□			
	Brook char	Δ	↑	A	□	□	□	□	□	□						
			↓	J						□	□	□	□			
	Brown trout	Δ	↑	A	□	□	□	■	■	■						
			↓	J	□	□	□	□	□	□	□	□	□	□	□	□
	Chinook salmon	Δ	↑	A	□	■	■	■	■	■						□
			↓	J	■	□	□	□	□	□	□	□	□	■	■	■
	Rainbow trout	Δ	↑	A				□	□	□	■	■	■	■	■	□
			↓	J	□	□	□	□	□	□	□	□	□	□	□	□
	Sockeye salmon	Δ	↑	A			□	■	□							
			↓	J					□	■	□					

Table A-2: Locomotory classification of some New Zealand freshwater fish species. Modified from Mitchell and Boubée (1989). Lamprey photo credit: Jane Kitson.

Mode of swimming	Species
Swimmers: Species that usually swim around obstacles. They rely on areas of low water velocity to rest and reduce lactic acid build-up with intermittent 'burst' type anaerobic movements to get past high water velocity areas.	Īnanga, smelt, grey mullet and common bullies. 
Anguilliformes: These fish can worm their way through small spaces between stones or vegetation either in or out of the water. They can breathe atmospheric oxygen if their skin remains damp.	Shortfin and longfin eels 
Climbers: These species climb the wetted margins of waterfalls, rapids, and spillways. They 'stick' to the substrate using surface tension and can have roughened 'sucker like' pectoral and pelvic fins or even a sucking mouth (like lamprey).	Lamprey, elvers (juvenile eels), juvenile kōkopu and kōaro. Juvenile and adult redfin bullies and, to a limited extent, torrentfish.  
Jumpers: These species can leap using the waves at waterfalls and rapids. As water velocity increases it becomes energy saving for these fish to jump over the obstacle.	Trout and salmon. 

Swimming performance is a critical factor determining the ability of fish to migrate and overcome barriers to migration. Swimming abilities can be used to determine water velocity conditions that need to be met for fish to pass over or through an instream structure. Swimming performance is typically defined in terms of the duration of swimming and the intensity (i.e., speed) at which the fish swims. There are three dominant swimming modes accepted by most researchers: (1) sustained swimming, (2) prolonged swimming, and (3) burst swimming (Beamish 1978; Hammer 1995; Kieffer 2010). Sustained swimming is aerobic, can be maintained for extended periods of time (typically >200 min) and does not involve fatigue. The prolonged swimming mode lasts between 20 seconds and 200 minutes and, depending on the swimming speed, ends in exhaustion. Burst swimming represents a form of high intensity, short duration (<20 secs), anaerobic activity (Beamish 1978). While the endurance thresholds between swimming modes have been widely cited, they are somewhat arbitrary and there is evidence to suggest that these thresholds vary between fish species and possibly individuals (e.g. Nikora et al. 2003).

Knowledge of swimming speeds in fish has advanced significantly over the last 50 years (Kieffer 2010; Katopodis and Gervais 2012). Critical swimming speed (Brett 1964) is the most frequently used and easiest method to measure swimming performance (Plaut 2001). It is essentially a measure of the prolonged swimming mode, with fish incrementally exposed to higher water velocities for a set period until they reach fatigue. Critical swimming speeds have frequently been used to inform the development of water velocity design criteria for providing fish passage at instream structures (Katopodis and Gervais 2012), although not without criticism (Peake 2004). Another commonly used measure of swimming performance is endurance, which provides information on how far and/or how long a fish can swim against a given water velocity (Brett 1964; Beamish 1978; Katopodis and Gervais 2012). These data have also been used to help inform design criteria for fish passage (e.g. Peake 2004; Laborde et al. 2016; Crawford et al. In review).

More recently, research has begun to focus on assessments of voluntary fish swimming performance in open channels, e.g., streams (Katopodis and Gervais 2012; Vowles et al. 2013). This has been facilitated by the emergence of biotelemetry methods that allow real-time tracking of fish movements and upstream progress allowing an assessment of swimming performance in real-world instream conditions (Haro et al. 2004; Goerig et al. 2016). Unfortunately, the utility of these techniques for many New Zealand species are limited by their small body size at migration (Franklin and Gee 2019). There is increasing evidence emerging that volitional swimming performance can be significantly different to that assessed under some controlled laboratory conditions (Peake 2004; Castro-Santos 2005; Mahlum et al. 2014; Goerig et al. 2016), likely reflecting the influence of natural environmental heterogeneity (e.g., turbulence and boundary layer conditions) and the impacts of fish motivation and behaviour on overall fish swimming performance.

Several studies have demonstrated the influence of environmental factors on fish swimming performance, including water temperature (Beamish 1978; Rodgers et al. 2014; Crawford et al. 2024), dissolved oxygen (Farrell et al. 1998; Landman et al. 2005) and turbulence (Enders et al. 2003; Nikora et al. 2003; Liao 2007; Silva et al. 2012), but understanding of these influences is still relatively poor in most cases, especially for New Zealand's native fish species. Physiological (e.g., age and fatigue) and behavioural (e.g., learning) factors are also thought to have an impact on fish swimming performance (Farrell et al. 1998; Liao 2007; Nyqvist et al. 2024), but remain relatively poorly studied (Kieffer 2010; Katopodis and Gervais 2012; Vowles et al. 2013).

Fish swimming ability increases with size (Bainbridge 1958; Nikora et al. 2003). Given that the majority of New Zealand's native fish species migrate upstream at a small size, they require more conservative design criteria for ensuring fish passage compared to salmonids (which migrate upstream as adults) and many of the other species that have been more widely studied in the Northern hemisphere. Crawford et al. (In review) quantified the swimming capabilities of nine migratory fishes from New Zealand finding significant inter- and intra-species variation. Īnanga have the lowest median absolute swimming speed (Figure A-3). Longfin elvers, redfin and common bullies and banded kōkopu have similarly poor swimming speeds. Shortjaw kōkopu, kōaro, smelt and giant kōkopu are all relatively stronger swimmers, but there is considerable inter-species variation in swimming capability that is important to consider in fish passage design (Figure A-3).

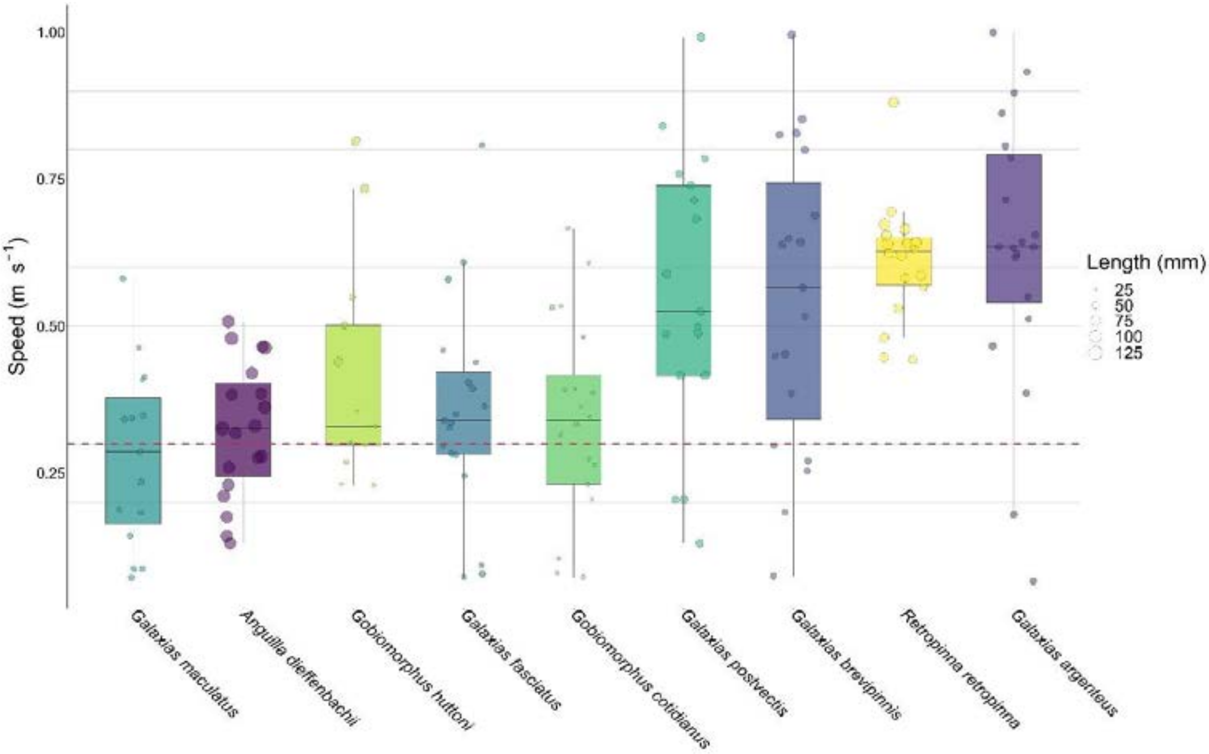


Figure A-3: Absolute swimming speeds of nine New Zealand freshwater fishes. Points represent individual fish. Size of each point represents the size of the fish. The box indicates the interquartile range, whiskers extend 1.5 × the interquartile range, and the centre bar indicates the median. Adapted from Crawford et al. (In review).

Crawford et al. (In review) developed fish passage design curves for īnanga as the weakest swimming species (Figure A-4). The design curves indicate the maximum allowable water speeds that will provide passage for different proportions of individual īnanga (50%, 70%, 90%) for different culvert lengths. Traditionally, most design curves are based on the median swimming speed of fish, meaning that the design velocities will exceed the swimming capabilities of 50% of fish. To provide passage for a higher proportion of individuals, Crawford et al. (In review) have demonstrated that considerably lower design water speeds are required. For example, the maximum allowable water speed that will provide for passage of 50% of fish through a 20 m culvert is approximately 0.43 m s^{-1}, but this drops to approximately 0.1 m s^{-1} to allow passage for 90% of individuals. It is important to note that the maximum allowable water speeds required to provide passage for 50, 70 or 90% of individuals are lower than the historical 0.3 m s^{-1} rule-of-thumb at culvert lengths greater than 40 m, 18 m, and 12 m respectively.

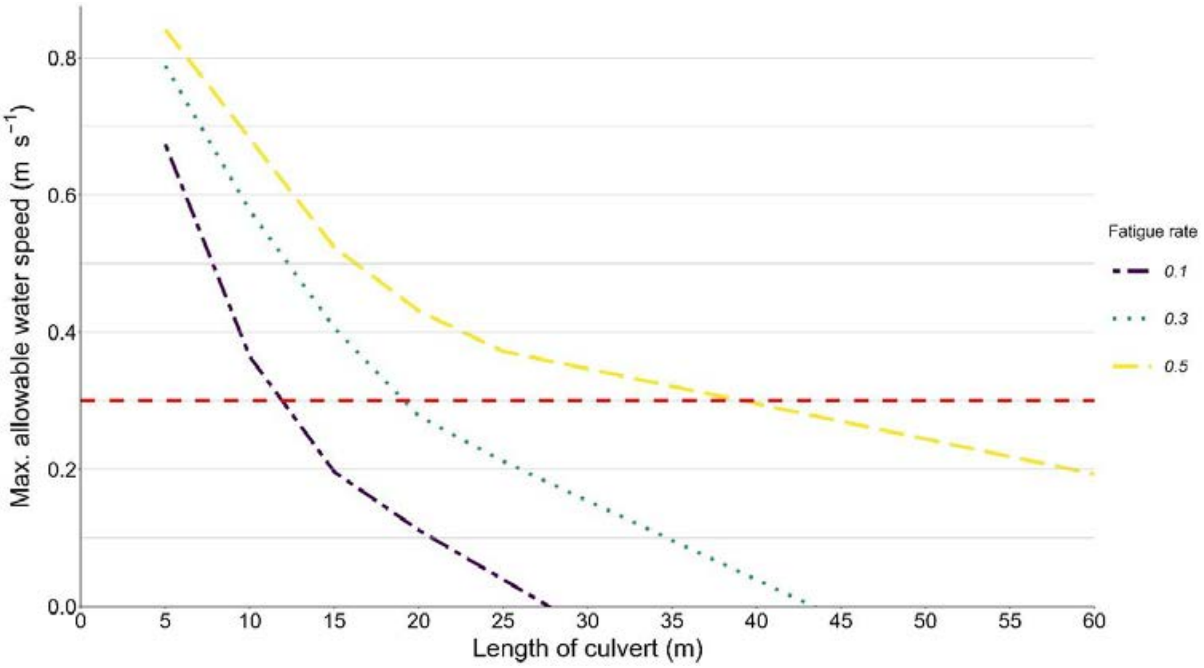


Figure A-4: Fish passage design curves for īnanga. The yellow dashed line represents the median design curve (i.e., will provide passage for 50% of fish). The green dotted line represents the design curve that will provide passage for 70% of individuals. The purple (dot dash line) represents the maximum allowable water velocities that will provide passage for 90% of individuals. The red dashed horizontal line represents the traditional rule-of-thumb design velocity of 0.3 m s^{-1} Source: Crawford et al. (In review).

While there is little published data on the climbing abilities of New Zealand freshwater fishes, certain species of New Zealand freshwater fishes have well-developed climbing skills. Among the galaxiids, banded kōkopu and kōaro are extraordinarily skilled climbers, and can pass significant falls (McDowall 2000). Galaxiids climb by unilateral pectoral fin movement, leading to a wiggling motion from side to side as they ascend. By contrast, bullies that can climb use a bilateral motion of both pectoral fins simultaneously to detach and re-attach to the wetted surface, climbing by little hops upwards. While common bullies are not known to be climbers, redfin bullies can surmount significant barriers by climbing, and bluegill bullies can pass moderate barriers (McDowall 2000). Shortfin and longfin elvers are also skilled climbers, longfins reputedly more-so than shortfins (McDowall 2000). Elvers climb by attaching themselves to the substrate using friction and surface tension, and undulating their bodies in an anguilliform motion as when swimming, but with their bodies adpressed closely to the substrate (Jellyman 1977). They often take advantage of rough substrate by wiggling between raised areas to provide greater surface area for adhesion. However, their ability to climb vertical surfaces is largely limited to individuals that are <120 mm in length (Jellyman 1977).

Appendix B What creates a barrier to fish movements?

Barriers to fish movements can be caused by natural and artificial features in streams (Franklin et al. 2014). Natural features such as waterfalls, cascades or naturally intermittently dry stream reaches can impede or prevent the movement of fish. However, as naturally occurring features, the impacts of natural barriers on stream communities are generally of little ecological concern and should not be removed or changed. The exception is when natural barriers provide protection for critical habitats or native fish populations, and/or constrain the spread of undesirable or exotic species. In such cases it is critical that these benefits are taken into consideration when developing a barrier management strategy (see Section 6 for further details).

Artificial structures, such as dams, culverts, weirs, and fords, can obstruct fish movements if adequate consideration is not given to catering for these movements during structure design, installation, and maintenance. While these structures impede the movements of fish, this result is generally an unintended consequence of the design and they have been termed ‘unintentional barriers’ (Charters 2013). Avoiding the creation of new, and improving the mitigation of existing, artificial, unintentional barriers is one of the primary objectives of these guidelines (see Sections 4 and 5 respectively).

In some cases, exclusion barriers are constructed or maintained to prevent undesirable fish accessing certain areas (Charters 2013; Franklin et al. 2014). These ‘intentional barriers’ can be physical obstructions (e.g., perched culverts, overhangs, dams, screened water intakes), that are designed to exceed the undesirable fishes’ ability to negotiate the barrier, or non-physical (e.g., acoustic and air bubble barriers, electric fields and strobe lighting), which are intended to stimulate an avoidance response by exotic or all species (Charters 2013). Design considerations for these intentional exclusion barriers are presented in Section 6.

Overall, there are several key structural features that can result in fish movements being impeded that may be present in natural and artificial, and in unintentional and intentional barriers. The following sections highlight some of these features and explain how they contribute to impeding fish movements.

Fall height

Any instream configuration, whether natural or artificial, can become an insurmountable obstacle for fish if it causes a sudden change in the water surface or bed level (Figure B-1). In the case of an artificial structure (e.g., culvert), this situation may occur at installation, or develop because of subsequent erosion. The vertical distance between the water level of the structure and the water level of the stream below is generally used to define the fall height of the structure.

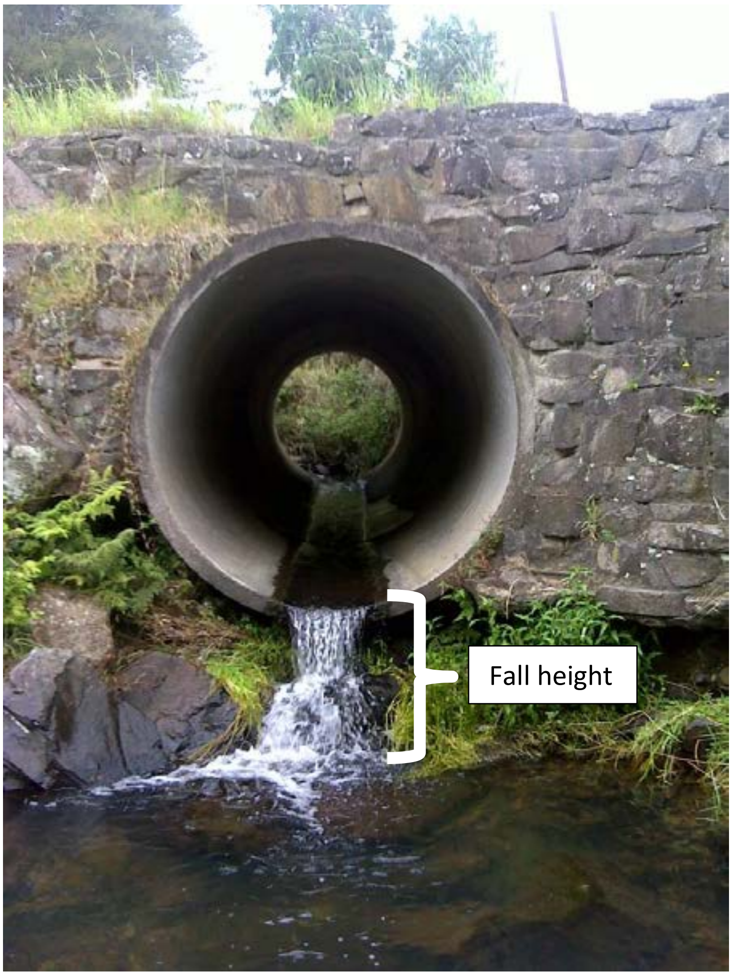


Figure B-1: Example of a perched culvert illustrating fall height.

The energy requirements for fish negotiating barriers increase with fall height, and the ability of different fish species to surpass obstacles will depend upon their individual swimming, climbing or jumping abilities, as well as their life stage. Baker (2003) examined the effect that the height of a weir may have on two migrating native fishes (the common bully and īnanga) that migrate by swimming. As fall height increased, the number of juvenile īnanga passing the weir decreased significantly, with only around 30% of fish passing the weir at 50 mm and none passing the weir with a 100 mm fall height (Figure B-2). For adult īnanga, the number of fish able to pass the weir as the fall height increased from 50 to 200 mm reduced rapidly from around 75% at 50 mm to no īnanga able to pass at the maximum fall height of 200 mm (Figure B-3). For adult īnanga, the size of the fish was significant in determining successful passage over the weir, with larger fish surmounting the weir with greater ease than smaller fish (Figure B-3). It is thought that the differences in fish passage ability between life stages of īnanga may be related to differences in muscle mass between juvenile fish (that had spent their lives in the sea and had relatively little muscle) and adults (who had been living in the river environment and had developed more musculature to cope with the flowing water they experience).

For common bullies, the number of bullies successfully passing the weir also decreased significantly as fall height increased. Again, fish size significantly influenced successful passage over the weirs with larger fish surmounting the weirs with greater ease than smaller fish (Figure B-4). No small common bullies passed the weir when the fall height was 100 mm or more and only 40% were able to pass at a fall height of 25 mm. Around 80% of large bullies could pass the 25 mm fall height, but this was reduced to 40% at 75 mm and zero at 125 mm (Figure B-4).

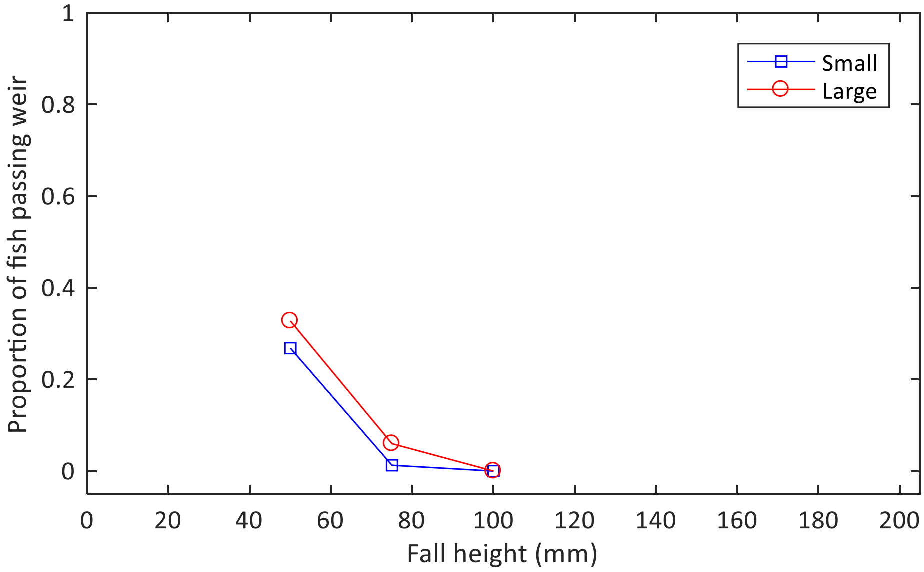


Figure B-2: Proportion of juvenile īnanga that passed a V-notch weir at different fall heights. 'small' = average size of 47 mm; range 45–49 mm. 'large' = average size of 51 mm; range 50–59 mm. Reproduced from Baker (2003).

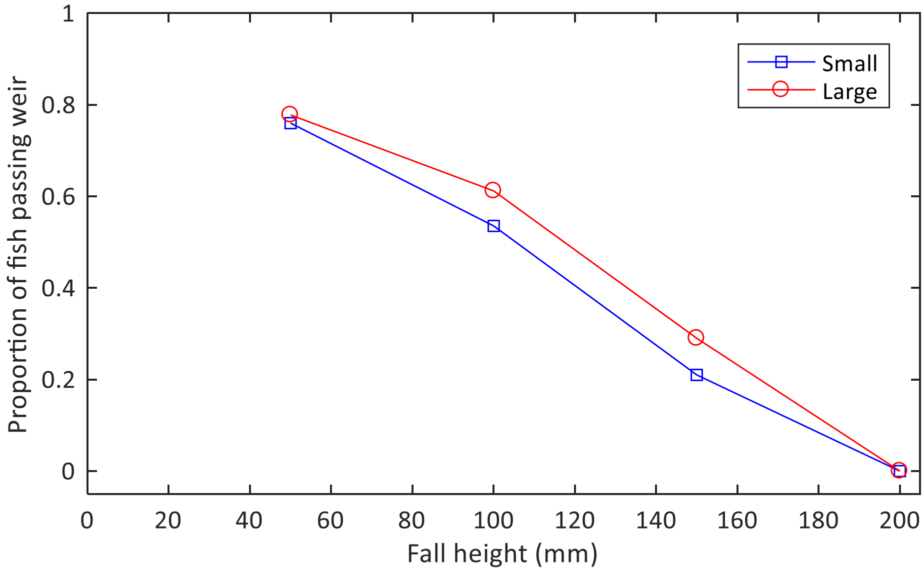


Figure B-3: Proportion of adult īnanga that passed a V-notch weir at different fall heights. 'small' = average size of 55 mm; range 44–60 mm. 'large' = average size of 66 mm; range 61–110 mm. Reproduced from Baker (2003).

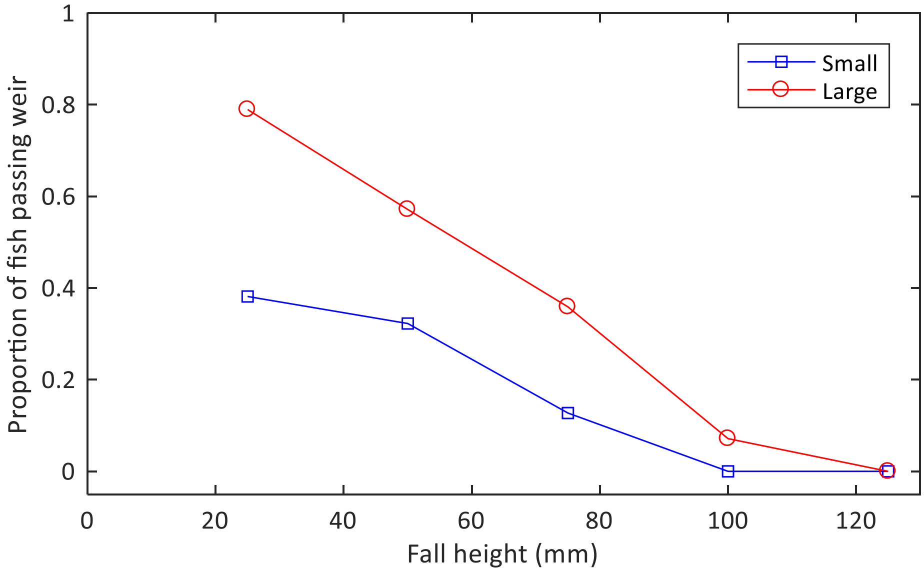


Figure B-4: Proportion of common bullies that passed a V-notch weir at different fall heights. 'small' = average size of 40 mm; range 28–50 mm. 'large' = average size of 57 mm; range 51–95 mm. Reproduced from Baker (2003).

Most climbing species can overcome significant fall heights, as long as there is a continuous wetted surface available for them to climb. However, where structures become undercut and an overhang develops, even climbers are unable to successfully pass. It is unclear what the potential energetic costs of climbing are when compared to swimming.

There is a lack of information about the jumping abilities of native species. However, the introduced brown trout (*Salmo trutta*) is known to traverse falls of at least 40 cm by jumping (Holthe et al. 2005) and brook trout (*Salvelinus fontinalis*) have been recorded jumping 74 cm (Kondratieff and Myrick 2006). Most of the research into fish jumping behaviours has been conducted on salmonids. Factors affecting the height of falls that can be jumped by salmonids include fish length and downstream pool depth (Brandt et al. 2005; Lauritzen et al. 2005; Kondratieff and Myrick 2006), water temperature (Symons 1978; Holthe et al. 2005) and upstream water velocity and turbulence (Stuart 1962 in Symons 1978). It is reasonable to assume that similar factors affect jump heights of other fishes. The ability to jump barriers means that small fall heights from culverts and weirs present less of an obstacle to upstream migration of brown trout and the other salmonids than to non-jumping species.

Water velocity

When water velocities exceed the swimming capability of fish, upstream migration will be prevented (Warren and Pardew 1998; Haro et al. 2004). This may occur around instream structures, naturally within the stream environment, or where channels have been modified (e.g., straightened, or artificial channels). The ability of a fish to overcome high water velocities is a function of their swimming capabilities, the distance over which they have to travel, whether low velocity refuge areas where they can rest and recover after swimming to exhaustion are present, and environmental conditions (Castro-Santos 2004; Peake 2004; Katopodis and Gervais 2012; Goerig et al. 2016). If water velocity restricts the distance a fish can travel at any one time to less than the full distance it needs to pass (e.g., Figure A-4), low velocity refuge areas will be required to allow fish to recuperate after bursts of swimming. However, even if a fish can maintain a stationary position between periods of forward movement, the energetic requirements to achieve this may mean that they become exhausted before they reach the end of the channel (Brett 1964; Enders et al. 2003). Furthermore, there may be a cumulative effect associated with the energy expended making multiple attempts to overcome a barrier, and/or in overcoming multiple barriers in sequence (Hinch and Rand 1998; Castro-Santos 2004).

To make upstream progress a fish must swim at a speed greater than the velocity of the water it is swimming in to (Peake 2004; Laborde et al. 2016). However, the duration for which a fish can maintain a given speed reduces as its swimming speed increases. Consequently, there is a trade-off between water velocity, swimming speed and the distance that can be travelled, and this must be considered when setting appropriate water velocity design criteria. However, it should be noted that this relationship will vary between individuals and species, with fish size, environmental conditions (e.g., water temperature) and the distance to be travelled. This variation is illustrated in the example in Figure B-5, which shows the range of average passable water velocities for īnanga of different sizes passing through culverts of different lengths and in the velocity design curves in Figure A-4.

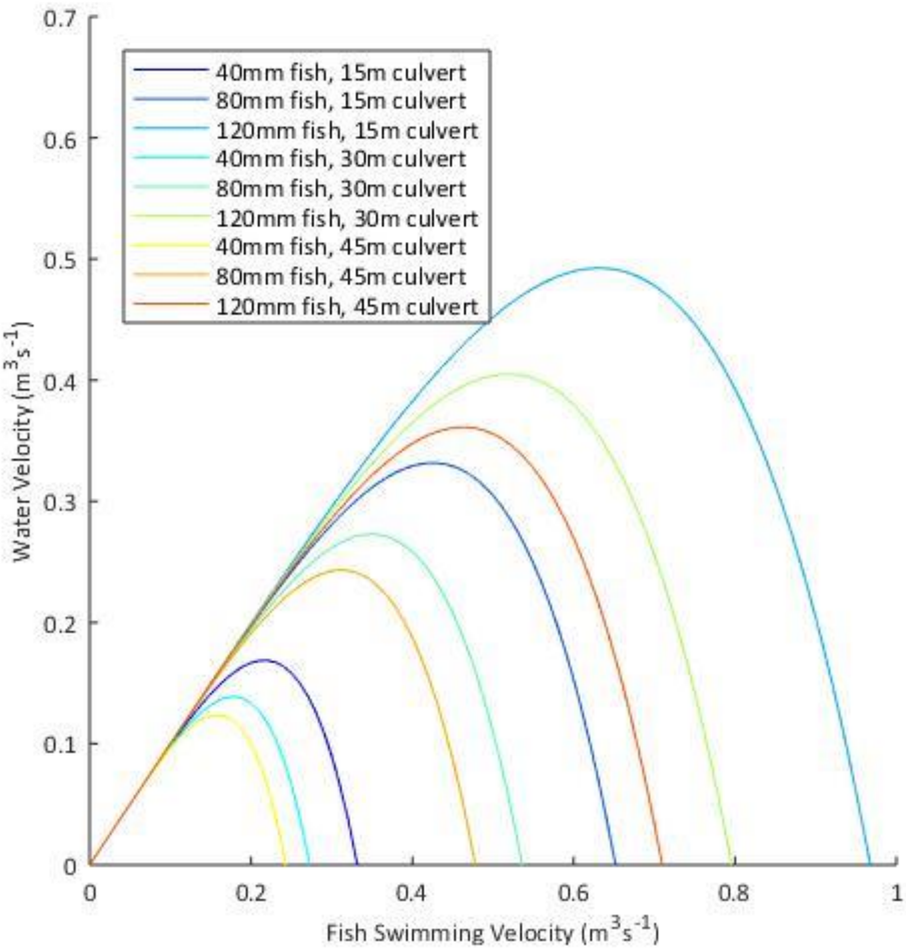


Figure B-5: Variation in passable water velocity for different sized īnanga in different culvert lengths.

It is also appropriate to recognise that in any situation there is spatial variability in water velocity and that fish are well adapted to exploiting these variations. For example, water velocity is always lower close to the bed or edges when compared to mid-stream, because of friction. Fish (particularly benthic species such as bullies) utilise these boundary layer conditions where water velocity is lower to facilitate their upstream movements. For example, this was suggested as a key driver for the greater passage success of salmonids through corrugated culverts as opposed to smooth culverts where the corrugations create a larger boundary layer with low velocity resting zones (Goerig et al. 2016).

Water depth

Insufficient water depth over or through structures can cause passage problems for fish. Shallow, flat aprons at the outlets of culverts or below weirs are an area where this commonly occurs (Figure B-6). Swimming ability is compromised for a partly submerged fish due to impacts on the efficiency of swimming (e.g., reduced thrust) and, if the gills are not fully submerged, reduced oxygen availability impacting aerobic performance (Webb 1975; Webb et al. 1991). Water depth design criteria are, therefore, typically defined based on the water depth required to fully submerge the target fish species and will be greatest where passage provisions are required for deeper bodied fish, e.g., adult kōkopu or trout.

In New Zealand, many upstream migrating fish species are small, can spend short periods out of water and have good climbing ability (McDowall 2000). Consequently, shallow water is not necessarily a problem for these fish and water depth could potentially be exploited as a means of limiting the movement of some of the larger exotic fish species present in New Zealand (see Section 6). However, in negotiating shallow water, fish are more susceptible to predation and the energetic implications of having to climb rather than swim are poorly understood (Kemp et al. 2009; McLaughlin et al. 2013).

Figure B-6: Example of shallow water that can act as a barrier to movement for fish. Photo credit: Eleanor Gee.

Turbulence

Most studies of fish swimming performance and locomotion have been carried out in a simplified hydrodynamic environment under uniform flow conditions (Liao 2007). However, such conditions are rare in nature and there is increasing evidence to show that fish swimming performance can be significantly altered in complex hydrodynamic environments (Enders et al. 2003; Liao et al. 2003a; Lupandin 2005; Silva et al. 2012).

When water flows over, through or around a structure, either natural (e.g., a rock) or artificial (e.g., a weir), velocity gradients are created that result in turbulent conditions of varying scales and intensities. Depending on the characteristics of turbulence in a given situation it can either attract or repel fish (Liao 2007). For example, there are numerous studies that document the increased energetic costs of swimming in turbulent flow (Hinch and Rand 1998; Enders et al. 2003; Tritico and Cotel 2010). However, turbulent flows that maintain an aspect of predictability can be exploited by fish to reduce the energetic costs of swimming (Liao et al. 2003b; Liao et al. 2003a). Other studies have demonstrated little difference in swimming performance between environments with uniform and turbulent flows, but acknowledge that this may be related to the scale of turbulent eddies relative to fish size (Nikora et al. 2003).

The differences in swimming performance between laboratory forced swimming experiments in controlled hydrodynamic conditions and volitional swimming behaviour in real-world situations are likely, in part, a consequence of fish exploiting natural hydrodynamic variability to facilitate upstream movements (Vowles et al. 2013). Large eddies (relative to body size) can provide low velocity resting areas, and the boundary layer close to the stream substrate also offers conditions that fish can exploit to save energy and improve passage rates. However, where structures create turbulence that elicits avoidance behaviour or that exceeds the swimming performance of fish, it can impede the passage of fish (Williams et al. 2012).

Physical blockage

Structures such as weirs, dams, tide gates and pumping stations can physically block the movement of fish, both upstream and downstream, by blocking streams and rivers. Jellyman and Harding (2012) showed that large dams alter freshwater fish communities in New Zealand by blocking fish migrations, with sites above dams having lower species richness, a lower percentage of diadromous species, and a higher percentage of exotic fish species, when compared to below dams. Weirs also often act as a temporal barrier to fish migration, with passage dependent on flow conditions overcoming the blockage caused by the weir (Winter and Van Densen 2001; Keller et al. 2012) (Figure B-7).

Figure B-7: An intake weir that blocks fish migrations on the Te Arai River near Gisborne. Photo credit: Jamie Foxley.

Doehring et al. (2011a) found that tide gates act as a temporal barrier to upstream migration of īnanga, with more than twice the number of fish passing an un-gated culvert than a culvert with a tide gate (e.g., Figure B-8). Delays in upstream migration were also observed at the gated site, with fish primarily moving upstream during high tide at the un-gated site, but having to wait until low tide when the gate was open at the gated site. Bocker (2015) found a significant increase in the number of native fish (īnanga and bullies) able to pass upstream through a tide gate when it was fitted with a fish friendly gate design. Bocker (2015) found that upstream migration of īnanga primarily occurred on the incoming tide, which is also when tide gates are closed.

The installation of fish friendly gate designs resulted in the gates remaining open for a longer period, including during the early phase of the flood tide. This allowed more fish to pass upstream with a twenty-fold increase in the number of whitebait captured upstream of the tide gate when the fish friendly gate was operating. Similar results have been observed in overseas studies, with Mouton et al. (2011) showing European glass eels blocked by a tidal barrier and Wright et al. (2016) reporting significant delays in upstream passage of adult brown trout at a tide gate.

Figure B-8: An example of a tide gate from the Waikato River catchment. Photo credit: Rimutere Wharakura.

Crest shape

The shape of a weir's crest has also been shown to impact on the ability of fish to pass. Baker (2003) investigated the effect of notch shape on fish passage over an experimental weir at varying fall heights (Figure B-9). It was shown that while notch shape had relatively little effect on the passage of īnanga, it did have a significant effect on the passage of common bullies under the test conditions. The optimal notch shape under the conditions tested by Baker (2003) was a v-notch design, with the least effective design being a wide rectangular notch. The differences in performance were attributed to the availability of low velocity margins on the edges of the channel that allowed fish to approach the weir before seeking out the high velocity flow at the base of the weir.

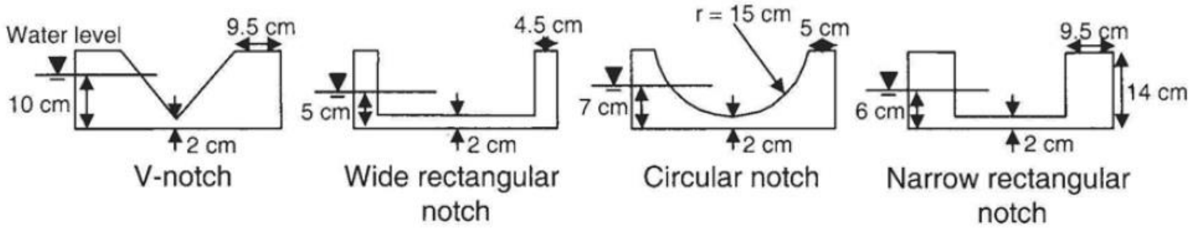


Figure B-9: Weir notch lateral cross-section shapes tested by Baker (2003).

A weir's longitudinal profile also impacts on the ability of climbing fish species to pass. Overhanging weir crests or weir crests with sharp (e.g., 90°) angles are more difficult for fish to pass than weir crests with a rounded profile (Figure B-10).

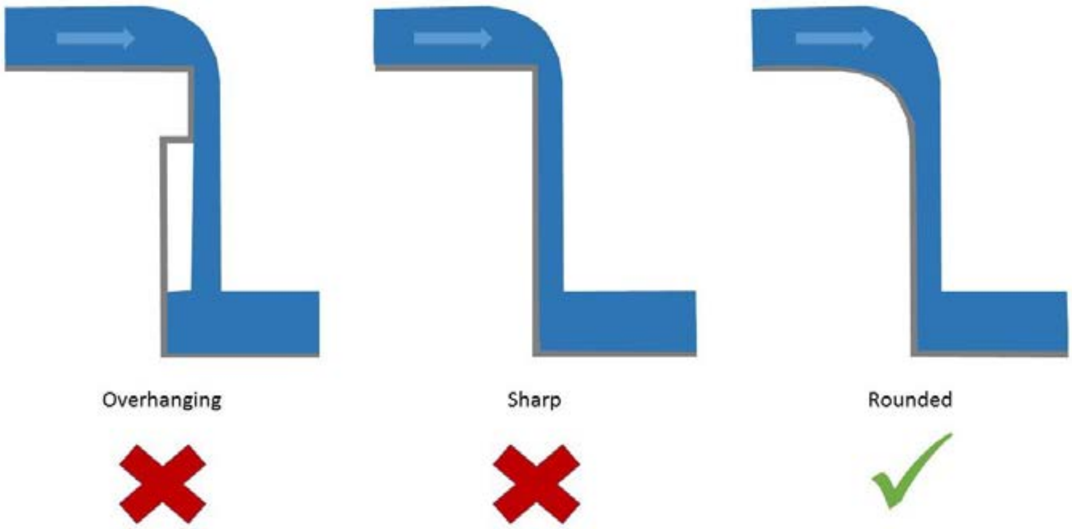


Figure B-10: Examples of different weir longitudinal cross-sectional profiles that influence fish passage success.

Silva et al. (2016) have demonstrated that the inclination of the upstream face of a spillway or weir impacts on downstream passage success of fish. They evaluated downstream passage success of the European eel (*Anguilla anguilla*) and Iberian barbel (*Luciobarbus bocagei*), a cyprinid species, at weirs with 30°, 45° and 90° upstream inclinations (Figure B-11). Both species avoided the turbulent area immediately upstream of the 90° weir, resulting in lower passage success, particularly for eels. However, with the sloped weir faces, this turbulent area was eliminated resulting in enhanced passage.

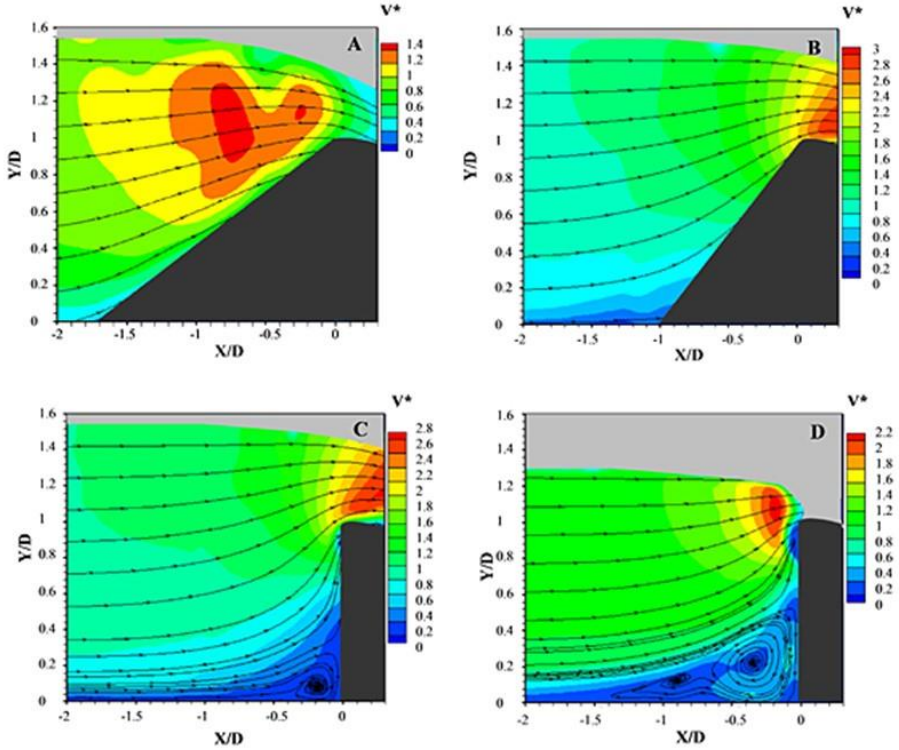


Figure B-11: Dimensionless water velocity (V*) and streamlines for the four experimental weir designs tested by Silva et al. (2016). Experiments conducted with depth of the approach flow H = 0.42 m and upstream face inclination of 30° (A), 45° (B) and 90° (C); and H = 0.32 m with upstream face inclination of 90° (D). Structures and areas outside the measured flow region are in dark and light grey, respectively. Dimensionless velocity V* values correspond to colours. Flow enters from the left. Source: River Research and Applications, Volume 32, Issue 5, pages 935–945 (http://onlinelibrary.wiley.com/doi/10.1002/rra.2904/full).

Attraction flows

Fish have an innate behaviour that leads them to orientate themselves into the flow (rheotaxis) (Arnold 1974). Rheotaxis is a multisensory behaviour in which the relative role of the different sensory cues is thought to vary with factors such as reference frame and proximity of objects (Baker and Montgomery 1999; Bak-Coleman et al. 2013; Elder and Coombs 2015). Rheotaxis behaviour is influenced by flow turbulence, and the presence of olfactory cues, and is a key behaviour driving migration.

During their upstream migration, fish are naturally drawn to conditions that indicate their migratory pathway will keep them within the main flow of a river (Williams et al. 2012). Instream structures typically alter flow pathways and hydraulic conditions, thus altering the cues for rheotaxis. Consequently, the flow conditions that a fish experiences at an instream structure are fundamental to achieving successful passage (Bunt et al. 2012). If appropriate flow conditions do not exist, fish will avoid, or fail to locate, the correct pathway upstream. This is a particular problem where only a small proportion of the flow is made available at a fish bypass, while most of the flow passes over or through a structure, e.g., at dams and weirs.

In this situation, sufficient attraction flow must be made available in the right configuration relative to the main flow to allow fish to locate the bypass and enter it without delay. Little work has been done with respect to attraction flow configuration for native species in New Zealand. O'Connor et al. (2015a) provides some guidelines on general principles of good attraction flow configuration.

During their downstream migration, eels effectively use a ‘reverse rheotaxis’ and actively seek out the dominant downstream flow pathways (Jellyman and Unwin 2017). Consequently, there is a challenge in ensuring adequate flow is provided to guide eels past instream structures. In contrast, the downstream dispersal of many of our other freshwater species during the larval life stage, e.g., galaxiids and bullies, is likely to be largely passive (e.g. Jarvis and Closs 2015).

Other factors

A range of other factors have been identified that may have an impact on passage success at an instream structure. Slope has been shown to influence passage success over ramps (Doehring et al. 2011b; Baker 2014). At a slope of 15°, īnanga and common bullies could pass ramps from 3 to 6 m in length, although passage success decreased with increasing ramp length. However, at 30° īnanga could only pass a 3 m ramp, and common bullies were incapable of passing any ramp length tested (Baker 2014). Passage success of redfin bullies was also reduced as ramp slope increased, but there was no significant effect of ramp length (Baker 2014). Doehring et al. (2011b) also found that as ramp angle increased from 5° to 20°, there was a significant reduction in the passage success of īnanga over a 3 m ramp with an artificial grass substrate. However, ramp slope has a significant effect on water velocity, with higher water velocities at higher slopes. It is, therefore, not clear to what extent the observed effect is a direct consequence of slope as opposed to greater water velocity or other hydrodynamic factors.

Light has also been proposed as having an effect on passage success, but there is limited evidence available to directly support this. Vowles and Kemp (2012) found that downstream migrating trout, which typically avoid sudden increases in water velocity, were extra-avoidant when light was present. Kemp et al. (2006) investigated the effects of light and dark conditions on downstream migrating salmon smolts passing a weir and found that different species and different sized conspecifics reacted differently in the presence or absence of light. The implications of these findings for upstream passage of juvenile fish in New Zealand are ambiguous, especially given the differences between species found by Kemp et al. A mark-recapture test of passage success of young-of-the-year *Galaxias spp.* in southern Australia through a 70 m long culvert found that passage success was unaffected by light conditions (Amtstaetter et al. 2017). However, in another Australian study, low light was shown to inhibit native fish movements through a vertical slot fishway suggesting that instream structures that alter light intensity may act as behavioural barriers to fish movement (Jones et al. 2017). In the same study, provision of artificial light of a similar intensity to daylight mitigated for the impact of reduced light.

Barotrauma, physical injuries caused by changes in water pressure, has been demonstrated as a cause of mortality in larval herring (Hoss and Blaxter 1979), and suggested as an explanation for higher mortality associated with fish passing undershot weirs than overshot weirs (Baumgartner et al. 2006). The construction or modification of structures should therefore avoid instigating conditions that lead to sharp changes in hydraulic head or water depth to minimise the risk of barotrauma to fish passing the structures.

Injury due to entrainment and/or impingement in flood control or irrigation pumps/intakes is a risk to migrating fish (Hickford et al. 2023); in New Zealand this is especially relevant to eels. Pump rotational speed appears to be a critical factor in rates of mortality, and grills over pump intakes during non-operational times may help prevent mortalities by excluding eels from sheltering in these spaces (Bloxham 2017). Large eels (> c. 600 mm) may suffer much higher mortality than smaller eels (Vaipuhi Consulting 2017). The type of pump impeller may also affect mortality. Downstream migrating European silver eels (*Anguilla L.*) suffered mortalities around 97% upon passage through a propeller pump, and around 17 – 19% when passing through an Archimedes screw pump (Buysse et al. 2014).

Bannon and Ling (2003) also demonstrated the potential consequences of degraded water quality on fish migrations through effects on fish swimming abilities. The sustained swimming abilities of juvenile rainbow trout, and larval and post-larval īnanga were shown to be compromised under elevated water temperatures and under mild hypoxia (75% dissolved oxygen saturation). This suggests that movement of migratory fishes through lowland rivers with degraded water quality could be significantly limited. In addition, point source discharges of pollutants can also alter migration patterns and heavy metals can render migratory fish unable to perceive odour and modify migration cues.

Examples of barriers

In practice, instream structures that are barriers to fish movements often combine several of the different features outlined above. The following pages contain examples of a range of obstructions to fish passage, with brief descriptions of why each constitutes a barrier. It is hoped that they will be instructive to those who are new to the topic of fish passage.

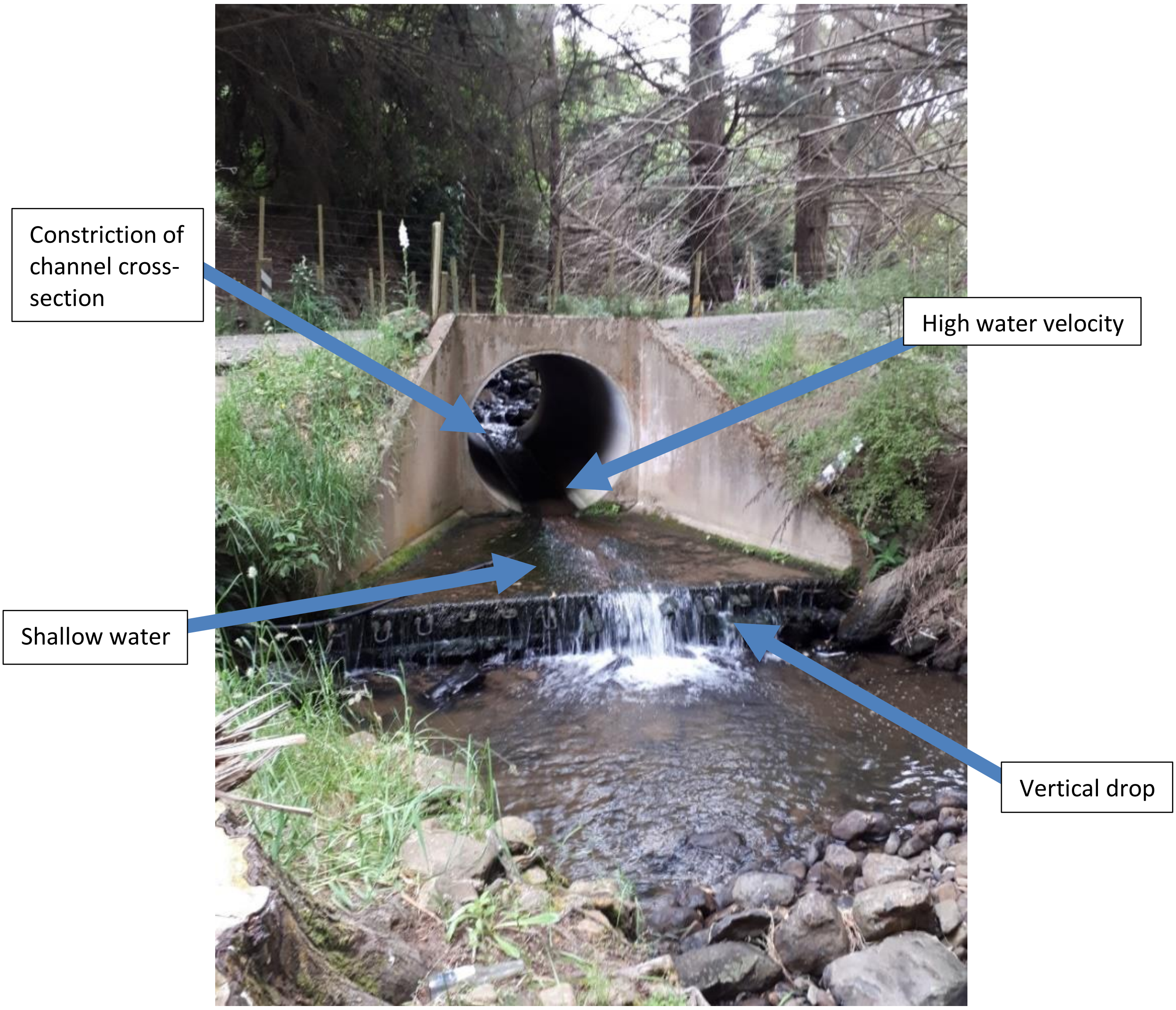


Figure B-12: Fish passage at this culvert will be impeded by the drop at the downstream end of the apron and the shallow and fast water on the apron and within the culvert. Photo credit: Sam Ammon.

Figure B-13: Example of a weir in central Christchurch. Fish passage will be impeded by the fall height of the steps in the weir and the salmonid fish passes. Photo credit: Megan Brown.

Figure B-14: Example of a culvert with a flap gate. Fish passage will be impeded by the flap gate being closed, even at low tide. Photo credit: Sam Ammon.

Figure B-15: Fish passage will be impeded by the fall height and undercut at the culvert outlet. The smooth culvert barrel will also lead to higher water velocities and limit fish movements under higher flows. Photo credit: Megan Brown.

Figure B-16: A double barrel culvert where passage will be impeded by the fall height at the culvert outlet. Passage for climbing fish species may be possible over the rocks below the culvert in the right of the picture. Photo credit: Sam Ammon.

Figure B-17: Fish movements will be impeded by the fall at the downstream end of the apron and shallow water on the apron. Photo credit: Sam Ammon.

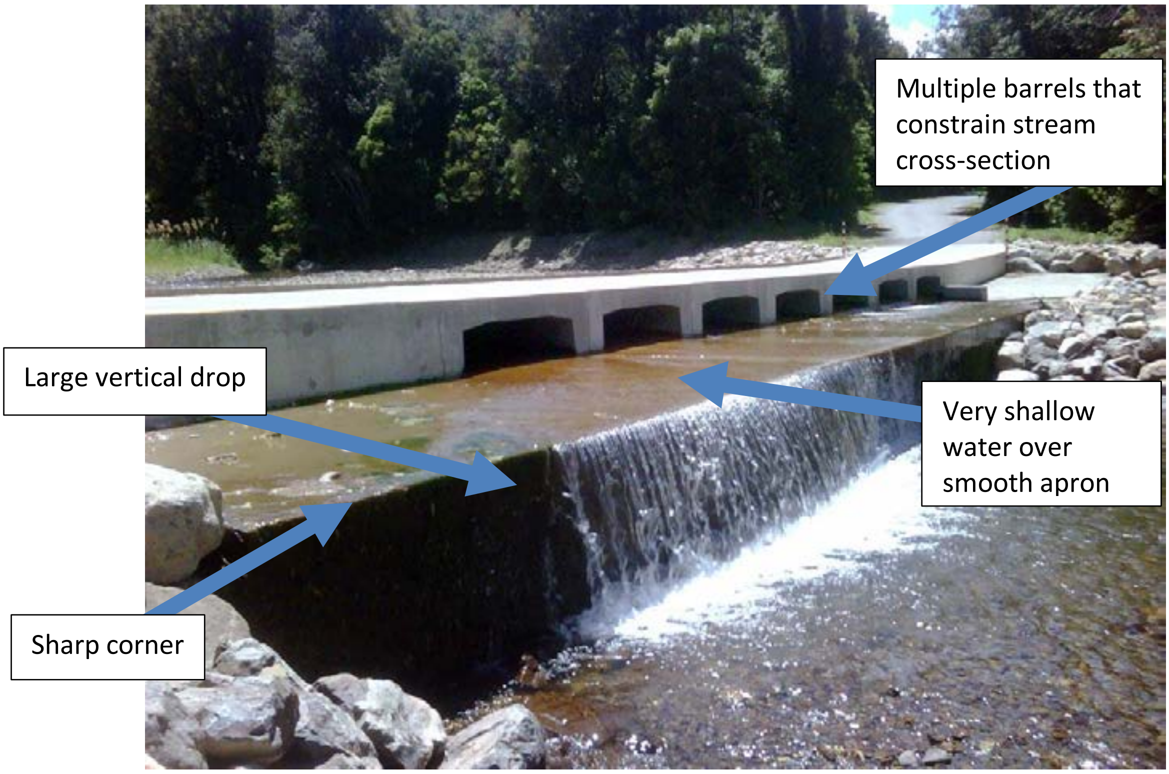


Figure B-18: An example of a drift-deck ford. This is a significant barrier to fish due to the large fall height on the downstream side of the structure. Passage may also be impeded by the sharp corner on the edge of the apron, shallow water, and high water velocities during high flow.

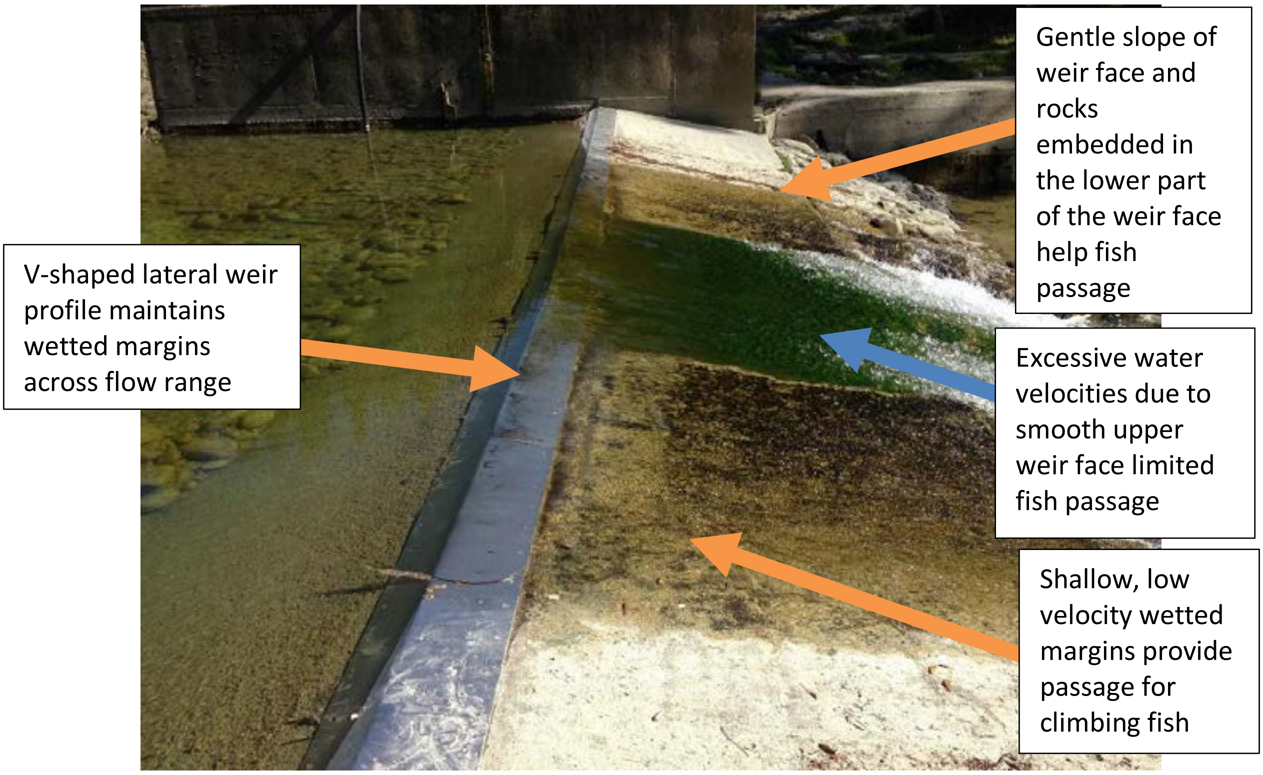


Figure B-19: An example of a crump weir used for hydrological gauging. Fish passage was impeded for swimming fish at this weir by the high water velocities. Climbing fish were able to use the wetted margins on each side of the weir to pass. Photo credit: Paul Franklin.

Figure B-20: Example of a ford from the Te Arai River. The fall height at the two steps on the downstream side of the ford makes a complete barrier to swimming fish species. Passage for climbing fish will be impeded by the sharp corners on the steps and shallow water across the ford, but some may access upstream habitats by taking advantage of the wetted margins to climb. Photo credit: Paul Franklin.

Figure B-21: The fall height on this ford on the Te Arai River is a significant impediment to the upstream passage of fish. Photo credit: Paul Franklin.

Figure B-22: Example of a culvert in a tidally influenced area. The culvert invert is embedded meaning that natural substrate is retained through the culvert. Culvert width relative to the stream bankfull width is lower than recommended, but because the culvert is in a tidal area, low water velocities will exist through the culvert during slack tide conditions. Photo credit: Bryn Quilter.

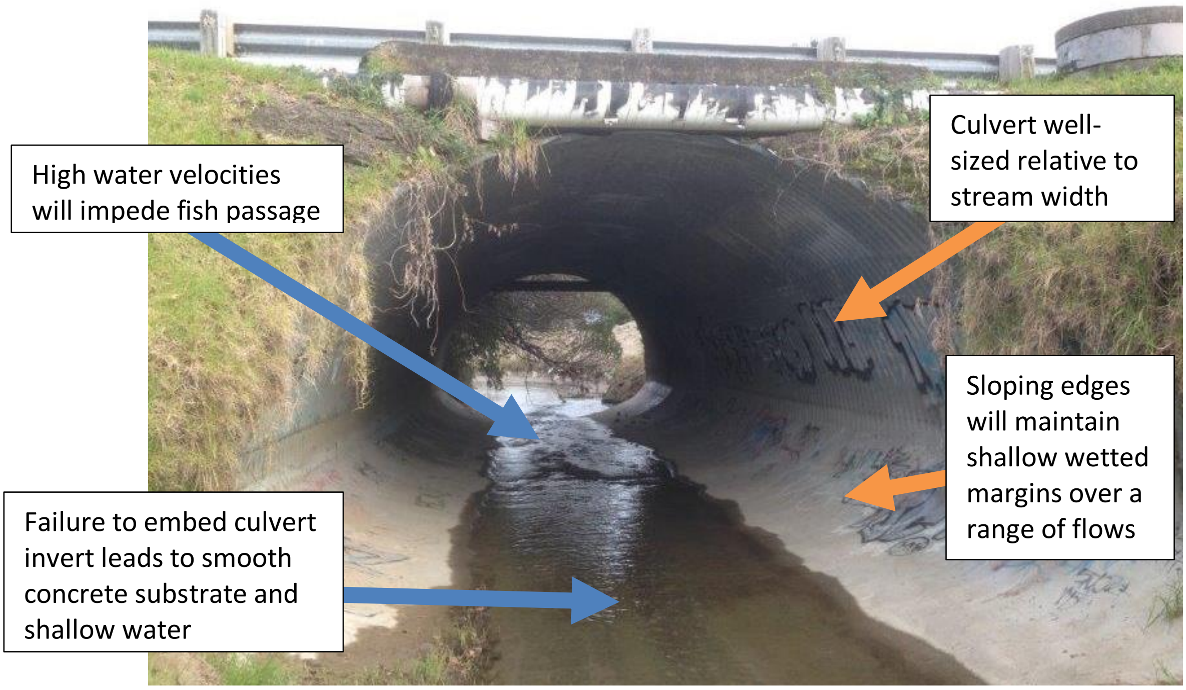


Figure B-23: Culvert well-sized relative to stream width, but failure to embed the culvert invert leads to shallow water depths and high water velocity. If this culvert had been embedded so that substrate was retained through the culvert it would have been a good example of how culverts can provide fish passage. Photo credit: Mark Pennington.

Appendix C Culvert design worked example 1 (HY-8 Version 7.80.2, Culvert Design)

Design procedure application.

The culvert evaluation software HY-8, developed by the United States Department of Transportation Federal Highway Administration (USDoT FHWA), incorporates a functionality that utilises the methodology presented by Zhai et al. (2014). This feature calculates the depth-averaged velocities within a vertical slice of the culvert based on the calculated cross-sectional average velocity within the culvert and the distance from the culvert wall. For evaluating fish passage, the software requires input regarding the upper and lower flows at which fish passage should be feasible. These flow parameters are essential for HY-8 to assess the hydraulic conditions and determine the potential for fish passage through the culvert.

Step 1. Hydrology

Determine design flows

The study site has undergone significant modification due to urbanisation. As a result, determining the bank-full flow is not appropriate. Therefore, the design flows for fish passage are determined as, fish passage upper threshold, $Q_{50\%\ 2\text{-year}}$, and fish passage lower threshold, $Q_{10\%\ 2\text{-year}}$. The EPA SWMM model is utilised to determine culvert and fish passage design flows. Table C-1 provides a summary of the design flow values computed by the EPA-SWMM model.

Table C-1: Design flows using SWWM model.

Fish Passage and Culvert Design Flows	$m^3\ s^{-1}$
$Q_{100-year\ total}$	16.59
$Q_{100-year\ \text{Barrel \#1}}$	9.34
$Q_{100-year\ \text{Barrel \#2}}$	7.25
$Q_{10-year\ total}$	6.54
$Q_{10-year\ \text{Barrel \#1}}$	3.78
$Q_{10-year\ \text{Barrel \#2}}$	2.76
$\mathrm{Q}_{50\%\ 2-\text{year total}}$	1.27
$\mathrm{Q}_{50\%\ 2-\text{year Barrel \#1}}$	0.90
$\mathrm{Q}_{50\%\ 2-\text{year Barrel \#2}}$	0.37
$\mathrm{Q}_{10\%\ 2-\text{year total}}$	0.25
$\mathrm{Q}_{10\%\ 2-\text{year Barrel \#1}}$	0.00
$\mathrm{Q}_{10\%\ 2-\text{year Barrel \#2}}$	0.25

Step 2. Stream characteristics

Identify reach characteristics

Tailwater Data

HY-8 provides several options for calculating the tailwater rating curve downstream from a culvert crossing. These options include rectangular, trapezoidal, and triangular channel shapes, as well as irregular channels, rating curves, and constant tailwater elevation. The 'Culvert Invert Data' option allows you to input the coordinates of existing culverts for analysis. The required input includes the inlet and outlet station and elevation, as well as the number of barrels.

In this example, the analysis of tailwater data is performed using an irregular channel.

The invert and outlet elevations for the culverts are as inlet elevation of 8.966 m and an outlet elevation of 8.466 m. Additionally, in this example, the culvert slope is computed by HY8 as 0.022 m/m for all barrels, and the culvert length is 22 m.

Step 3. Culvert design

Initial culvert design for larger design flows, including the 10-year and 100-year

Culvert diameter shall be selected that design peak flood flow can adequately pass according to the requirements of the Waka Kotahi NZ Transport Agency or guidelines/standards of the appropriate local council. The culvert configurations by conducting multiple trials are designed as a two-barrel circular culvert having a diameter of 2,050 mm. The Manning`s roughness is designed in HY8, 0.013 for the culvert wall, 0.065 for low flow and 0.046 for high flow for culvert bottom using Limerinos Equation. The culvert is designed with straight geometry, a 90-degree square edge, and a headwall with a k_e value of 0.5.

To input culvert data, select the 'Input Properties' option from the Culvert menu or right-click on the culvert in the Project Explorer window and choose Input Properties. Required culvert information includes culvert shape (e.g., circular pipe, box, elliptical long axis horizontal, pipe-arch, arch, low-profile arch, high-profile arch, metal box, concrete open-bottom arch, South Dakota concrete box and user defined), material (e.g., corrugated steel, steel structural plate, corrugated aluminium, aluminium structural plate, reinforced concrete, PVC, smooth HDPE and corrugated PE), size, and type (e.g., straight, side tapered, slope tapered, single broken-back and double broken-back).

Step 4. Culvert embedment design

Determine correct rock size or grain size and embedment depth

HEC-26 (Kilgore et al., 2010) includes a methodology to determine whether the embedment material included in the culvert design is stable under various flow rates. The methodology is based on estimating the shear stress required to move the embedment material and the shear stress present because of the flow rate through the culvert. HY-8 has incorporated this methodology as its Aquatic Organism Passage (AOP) analysis feature. Input data for AOP includes cross-sectional information obtained from the field describing the cross-sectional shape, thalweg elevation, slope (Figure C-1) and bed gradation (Figure C-2). Based on flow rates and the cross-sectional information, AOP computes Manning's n for each of the cross-sections and each of the flow rates, using four different methods. For each Manning's calculation, AOP determines whether the input parameters met the criteria of the method. It is up to the user to determine the most suitable method to be used for the conditions investigated. The final input information obtained in the field will determine if the streambed is in dynamic equilibrium (Figure C-2).

Based on the hydraulic input parameters for the culvert and the upstream and downstream channels, the AOP function in HY-8 calculates the applied shear stress to the bed, the permissible shear stress to the culvert bed and the minimum and maximum shear stress applied to the channel (Figure C-3). Calculations are carried out for high flow and peak design flows.

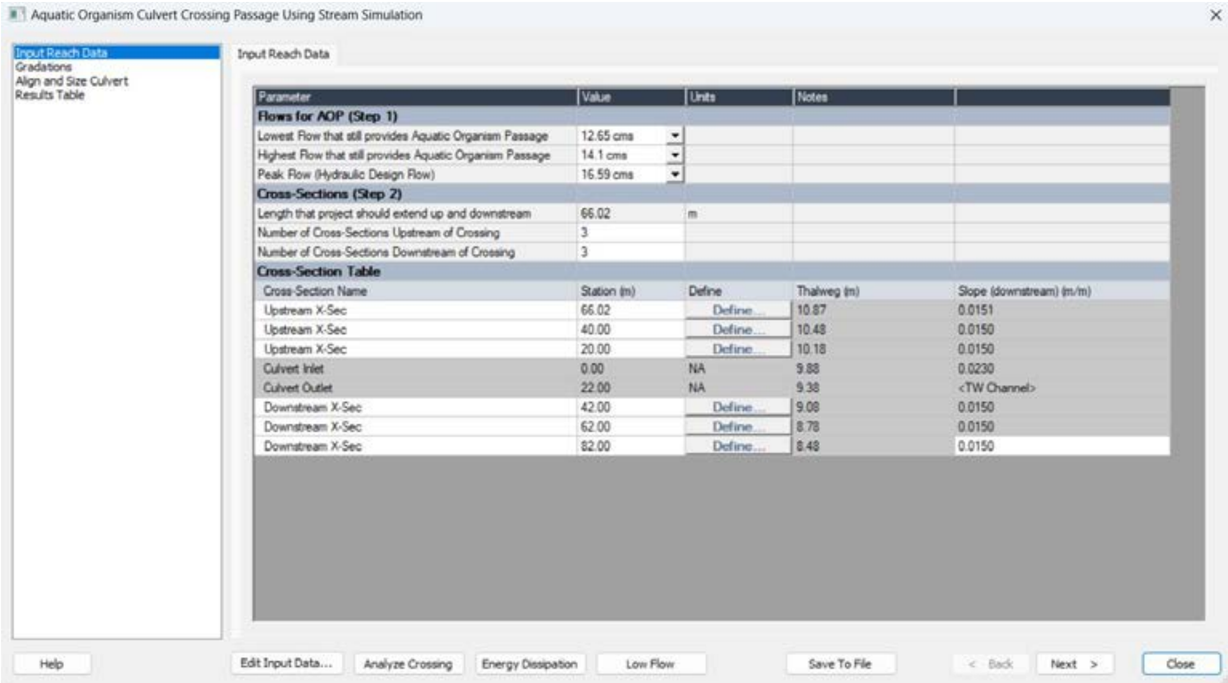


Figure C-1: **HY-8 AOP input reach data.**

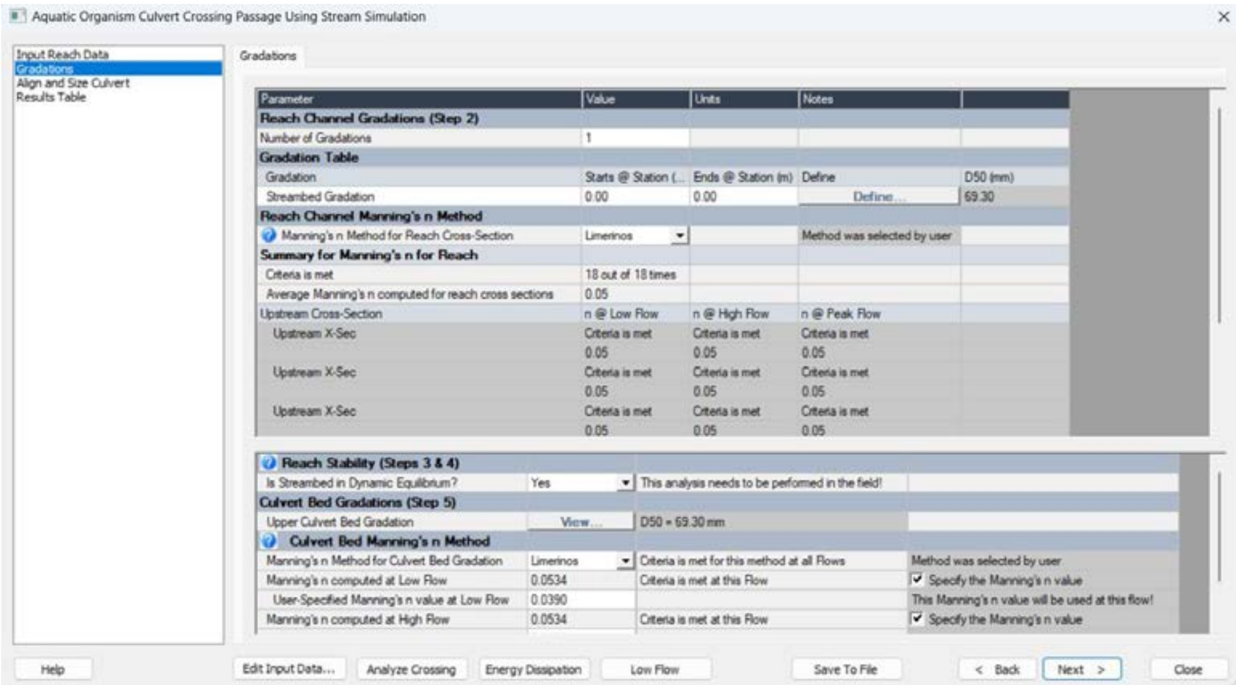
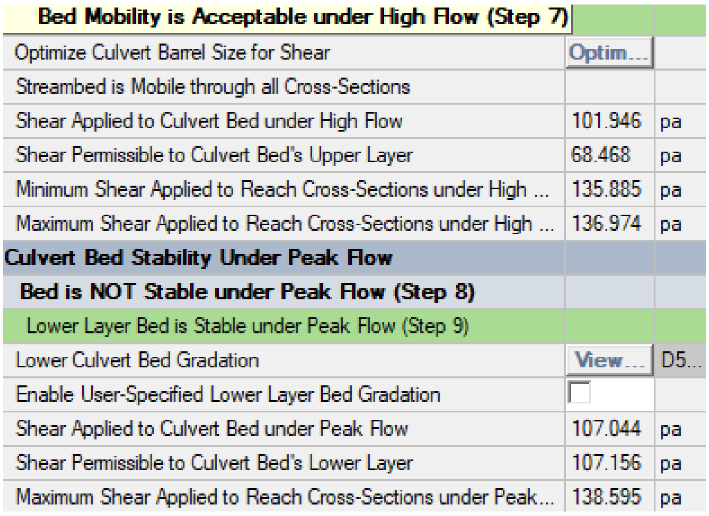


Figure C-2: **HY-8 AOP gradation input data.**



Bed Mobility is Acceptable under High Flow (Step 7)		
Optimize Culvert Barrel Size for Shear	Optim...	
Streambed is Mobile through all Cross-Sections		
Shear Applied to Culvert Bed under High Flow	101.946	pa
Shear Permissible to Culvert Bed's Upper Layer	68.468	pa
Minimum Shear Applied to Reach Cross-Sections under High ...	135.885	pa
Maximum Shear Applied to Reach Cross-Sections under High ...	136.974	pa
Culvert Bed Stability Under Peak Flow		
Bed is NOT Stable under Peak Flow (Step 8)		
Lower Layer Bed is Stable under Peak Flow (Step 9)		
Lower Culvert Bed Gradation	View...	D5...
Enable User-Specified Lower Layer Bed Gradation	☐	
Shear Applied to Culvert Bed under Peak Flow	107.044	pa
Shear Permissible to Culvert Bed's Lower Layer	107.156	pa
Maximum Shear Applied to Reach Cross-Sections under Peak...	138.595	pa

Figure C-3: HY-8 AOP align and size culvert.

Steps 5 and 6. Test for fish passage

Swimming velocity threshold and water depth

To analyse the velocity distribution and depth needed for fish passage, right click on the Barrel #1 culvert as presented in Figure C-4, then click on 'Low Flow' in the pop up window as depicted in Figure C-5.

To analyse the velocity distribution and depth required for fish passage, follow these steps:

- Right-click on 'The Barrel #1 Culvert' as shown in Figure C-4.
- In the pop-up window, click on 'Low Flow' as depicted in Figure C-5.

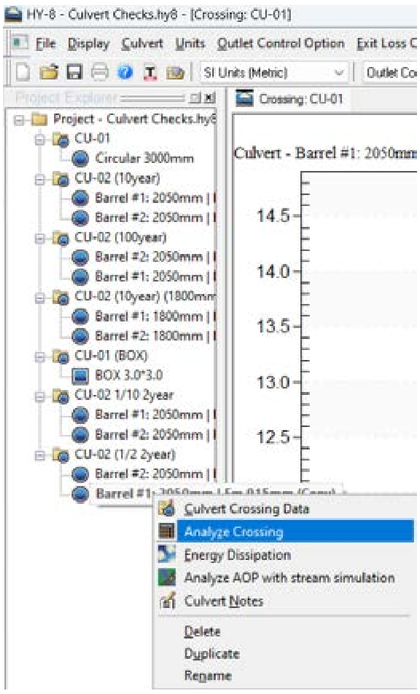


Figure C-4: HY-8 Analyse Crossing tool.

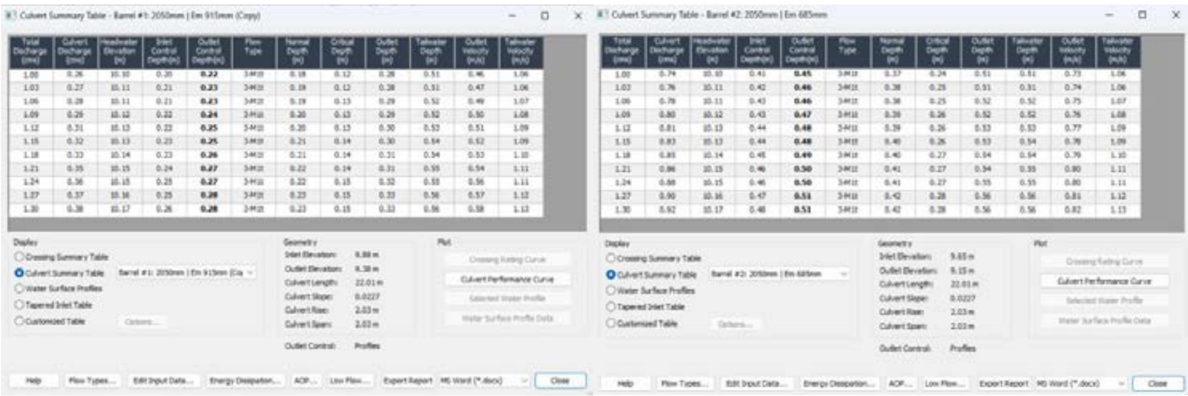
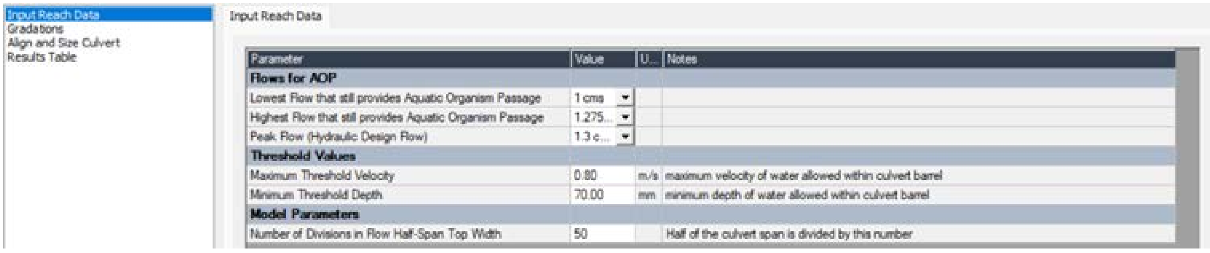


Figure C-5: HY-8 Low Flow tool.



Input Reach Data
Gradations
Align and Size Culvert
Results Table

Input Reach Data

Parameter	Value	U...	Notes
Flows for AOP			
Lowest Flow that still provides Aquatic Organism Passage	1 cms		
Highest Flow that still provides Aquatic Organism Passage	1.275...		
Peak Flow (Hydraulic Design Flow)	1.3 c...		
Threshold Values			
Maximum Threshold Velocity	0.80	m/s	maximum velocity of water allowed within culvert barrel
Minimum Threshold Depth	70.00	mm	minimum depth of water allowed within culvert barrel
Model Parameters			
Number of Divisions in Flow Half-Span Top Width	50		Half of the culvert span is divided by this number

Figure C-6: HY-8 Threshold Values.

The low flow hydraulics results for this particular example are shown in Figure C-8. The number of slices or division in flow half-span top width was set to 50 (Figure C-6). Based on the assumption that fish passage is required within the 0.15 m closest to the culvert wall, take the velocity of that slice as the culvert velocity.

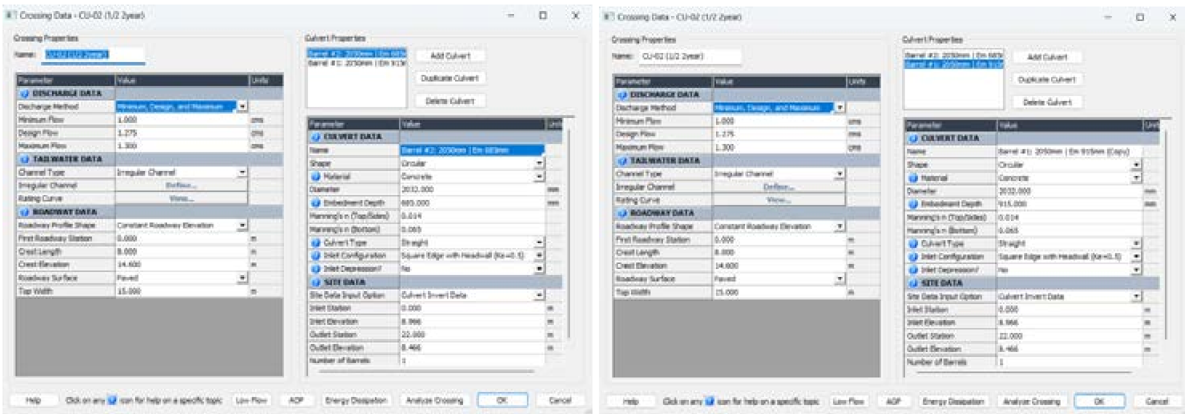
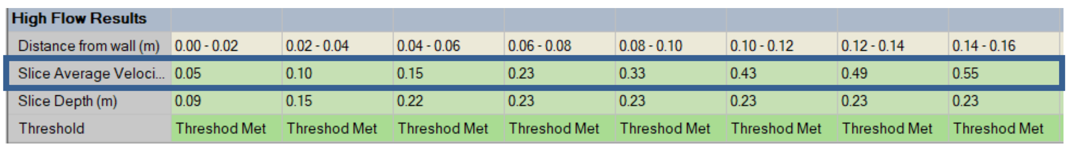


Figure C-7: HY-8 input parameters for evaluation of example culvert design including fish passage.



High Flow Results								
Distance from wall (m)	0.00 - 0.02	0.02 - 0.04	0.04 - 0.06	0.06 - 0.08	0.08 - 0.10	0.10 - 0.12	0.12 - 0.14	0.14 - 0.16
Slice Average Veloci...	0.05	0.10	0.15	0.23	0.33	0.43	0.49	0.55
Slice Depth (m)	0.09	0.15	0.22	0.23	0.23	0.23	0.23	0.23
Threshold	Threshold Met	Threshold Met	Threshold Met	Threshold Met	Threshold Met	Threshold Met	Threshold Met	Threshold Met

Figure C-8: Results for evaluation of example culvert design including fish passage in 50% of 2-year.

High Flow Results								
Distance from wall (m)	0.00 - 0.02	0.02 - 0.04	0.04 - 0.06	0.06 - 0.08	0.08 - 0.10	0.10 - 0.12	0.12 - 0.14	0.14 - 0.16
Slice Average Veloci...	0.04	0.09	0.13	0.19	0.28	0.36	0.42	0.47
Slice Depth (m)	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19
Threshold	Threshod Met	Threshod Met	Threshod Met	Threshod Met	Threshod Met	Threshod Met	Threshod Met	Threshod Met

Figure C-9: Results for evaluation of example culvert design including fish passage in 10% of 2-year.

As indicated in Figure C-8, the average velocity in a slice 150 mm distance from the wall, calculated as 0.31 m s^{-1} (average velocity of slices). Based on Figure C-10, the swimming endurance for the swimming velocity of 0.6 m s^{-1} assuming 75% success navigation for, the target species of New Zealand native fish, īnanga, is 95 s. Therefore, the effective velocity through the culvert is 0.29 m s^{-1} (0.6–0.31 m s^{-1}). The swimming distance can be calculated as 27.55 m (0.29 m $s^{-1} \times 95$ s), which is more than the culvert length 22 m. In accordance with Figure C-9, the depth (m) in a slice 150 mm distance from the wall is 190 mm. This exceeds the requirements of the fish passage which is 150 mm for the target species of New Zealand native fish, īnanga.

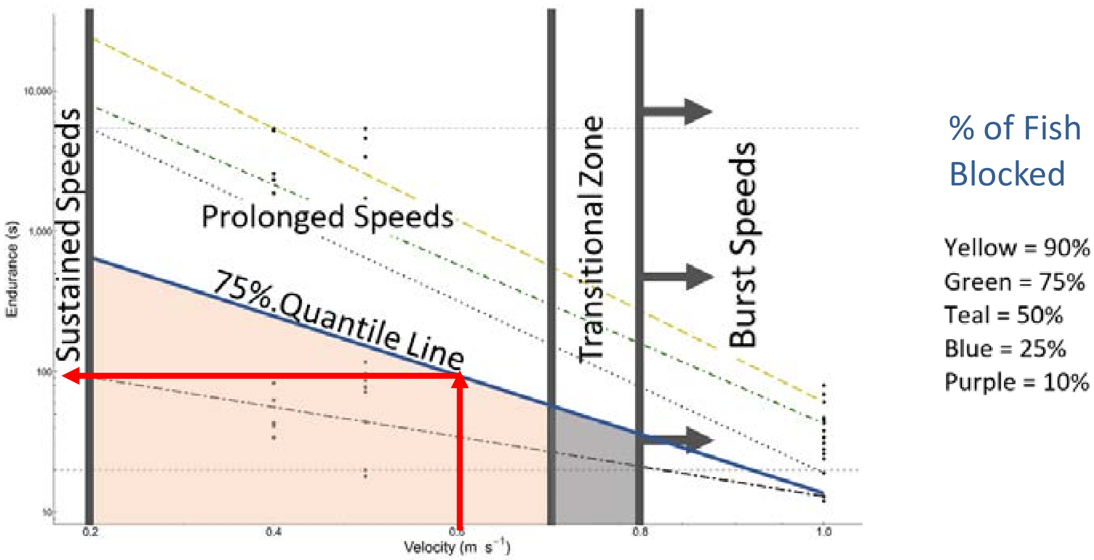
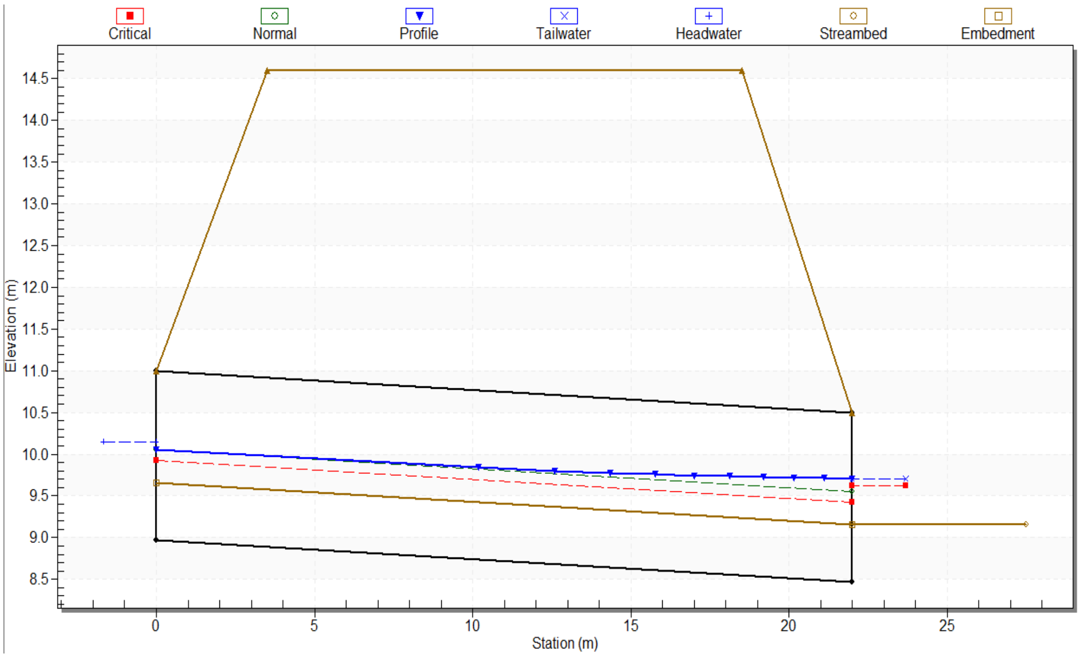


Figure C-10: Preliminary swimming speeds for the target species, īnanga.

Step 7. Design complete and ready for review

The culvert design parameters are established based on designated design flows: a peak flow (Q_{100}) of 16.59 m^3 s^{-1}, a High Fish Passage Flow ($Q_{50\%\ 2\text{-year}}$) of 1.27 m^3 s^{-1}, and a Low Fish Passage Flow ($Q_{10\%\ 2\text{-year}}$) of 0.25 m^3 s^{-1}. Essential design specifications include a diameter of 2,050 mm. Barrel #1 and Barrel #2 are embedded to a depth of 915 and 685 mm, respectively. The median grain size (D_{50}) of the materials utilised is 300 mm. For the $Q_{50\%\ 2\text{-year}}$ flow, the swimming distance exceeds the actual length of the culvert, ensuring fish passage. Additionally, during the $Q_{10\%\ 2\text{-year}}$, the minimum depth of water within the culvert is maintained above 150 mm, facilitating adequate conditions for aquatic life transit. Figure C-11 shows a detailed visual representation of the culvert's dimensions.



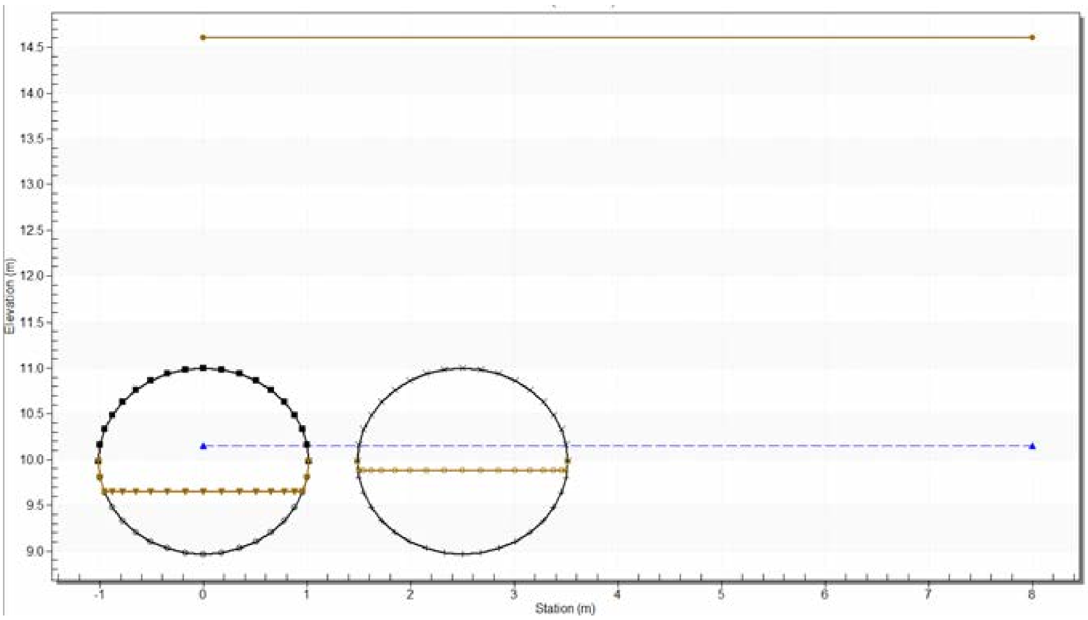


Figure C-11: Cross-section (bottom) and long-section (top) views of the two-barrel circular culverts.

Appendix D Culvert design worked example 2 (HEC-RAS 1D Version 6.3.1, Multi-Cell Round Culvert Design)

Design procedure application

The velocity for the fish passage design flow within the culvert is equal to one-third of the culvert's cross-sectional average velocity. The depth for the fish passage design should be based on the depth of 10% of a 2-year flow, which must be greater than 150 mm. Finally, the swimming distance should be more than the culvert length. For round culverts, a minimum embedment is between 1/3 to 1/2 of the culver diameter and for box culverts, the minimum embedment should be either 300 mm or 2 × D_{50}, whichever is greater.

Step 1. Hydrology

Determine design flows

The input parameters required for the HEC-RAS 1D steady model to facilitate fish passage include the design flows necessary for fish passage and the downstream normal depth, typically representing the downstream slope.

The study site has undergone significant modifications, due to urbanisation, therefore bank-full discharge is set to be 50% of a 2-year flow as fish passage upper threshold, ($Q_{50\%\ 2\text{-year}}$) and 10% of a 2-year flow is considered as fish passage lower threshold ($Q_{10\%\ 2\text{-year}}$). Furthermore, Design Culvert Flow, (Q_{100}) (corresponds to the flow that is equivalent to a 100-year ARI according to the requirements of the Waka Kotahi NZ Transport Agency or guidelines/standards of the appropriate local council. The design Fish Passage Flows, which have been calculated from the SWMM model, are summarised in Table D-1.

Table D-1: Design flows used in the HEC-RAS model.

Design Flows Fish Passage Flow	$m^3\ s^{-1}$
Q_{100}	2.36
$Q_{100\text{-year Culvert \#1}}$	1.64
$Q_{50\%\ 2\text{-year}}$	0.30
$Q_{50\%\ 2\text{-year Culvert \#1}}$	0.25
$Q_{10\%\ 2\text{-year}}$	0.06

Step 2. Stream characteristics

Identify reach characteristics

As stated in the guideline provided in HEC-26 (2010), the minimum extension of the project reach should be either 60 m or three culvert lengths, whichever is greater, from upstream and downstream of the crossing location. Given the existing culvert length of 35 m, the project reach must extend at least 105 m (three times the culvert lengths) from upstream and downstream of the culvert, as shown in Figure D-1. Additionally, the up and downstream channels are trapezoidal channels with a bottom width of 2 m and side slopes of 1(V):3(H).

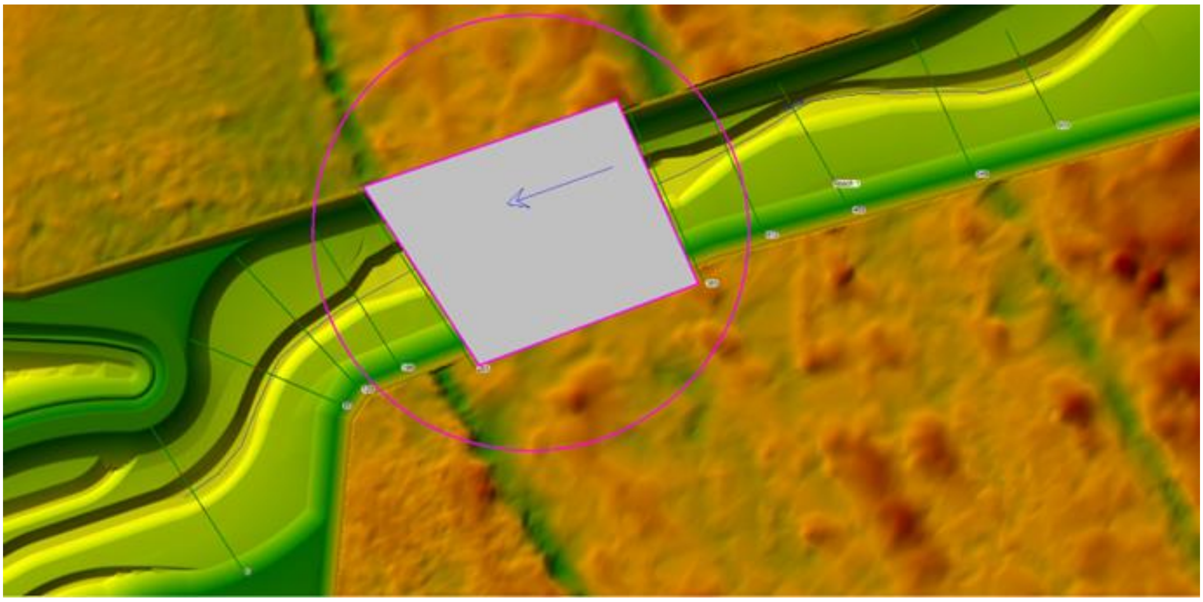


Figure D-1: Site reach and cross-section schematic.

Step 3. Culvert design

Initial culvert design for larger design flows, including the 10-year and 100-year

Culvert diameter shall be selected according to the requirements of the Waka Kotahi NZ Transport Agency or guidelines/standards of the appropriate local council. The culvert configurations by conducting multiple trials are designed as a multi-cell round culvert with a two-barrel round culvert of 3,000 mm in diameter. The Manning's roughness coefficients are 0.013 for the top and side walls, and 0.085 for the bottom.

Step 4. Culvert embedment design

Determine correct rock size or grain size and embedment depth

To assess the maximum shear stresses exerted on the bed material within the culvert, hydraulic analyses are conducted for 100-year, 10-year or a discharge which produces maximum shear stress. The applied shear stress for the culvert (τ) can be obtained using the formula:

$$\tau = \gamma \times y \times S_f \tag{20}$$

Where:

γ	= specific weight of water
y	= flow depth
S_f	= friction slope

For the primary culvert, Culvert #1, assuming 100-year flow produces maximum shear stress, the discharge is 1.64 $m^3\ s^{-1}$, the depth (y) is 0.74 m, the cross-sectional area (A) is 1.92 m^2, and the hydraulic radius (R) is calculated as 0.74 m, cross sectional area, divided by 4.08 m wetted perimeter. The manning roughness is calculated using manning roughness for multiple materials as 0.0467 from Equation (6). From Manning`s equation, the friction slope is:

$S_f \quad = (\frac{Qn}{AR^{2/3}})^2 = (\frac{1.64 \times 0.0467}{1.92 \times 0.47^{2/3}})^2 = 4.31 \times 10^{-3}$ m m^{-1}

Then:

$$\tau = 9810 \times 0.74 \times 4.31 \times 10^{-3} = 31.35 \text{ Pa}$$

The calculated shear stress in the culvert is 31.35 Pa. According to Table 4-2, very coarse gravel-fine cobble is the required embedded material for the culvert with D_{50} of 75 mm. For round culverts, the embedment depth can range from one-third of D to half of D. Thus, for the primary barrel, the embedment depth is calculated to be a third of the diameter, which is 1 m. For the secondary barrel, the embedment depth is calculated to be half of the diameter, resulting in 1.50 m.

Steps 5 and 6. Test for fish passage

Swimming velocity threshold and water depth

Taking into account that 25% of fish are unable to successfully navigate through the culvert, the endurance duration (s) for target species, at for example 0.6 m s^{-1} swimming speed, can be extracted from Figure D-2.

For half of the 2-year event, the average velocity at the inlet cross-section of Culvert #2 was calculated to be 0.08 m s^{-1}. The adjusted velocity, considering one-third of the measured velocity, was determined to be 0.027 m s^{-1}. Based on the swimming velocity of the targeted fish species, īnanga, at 0.6 m s^{-1}, the effective velocity (V_{ef}) was calculated to be 0.573 m s^{-1}, the swimming duration data from Figure D-2, is a duration (t_p) of 95 s. This results in a swimming distance of 54.5 m when applying the formula (0.57 m s^{-1}) × 95 s. Comparing this to the culvert length of 35 m, it is evident that the fish can navigate a distance greater than the length of the culvert. Therefore, it is concluded that the target fish species, īnanga, can adequately navigate through Culvert #2, given that the swimming distance exceeds the culvert's length.

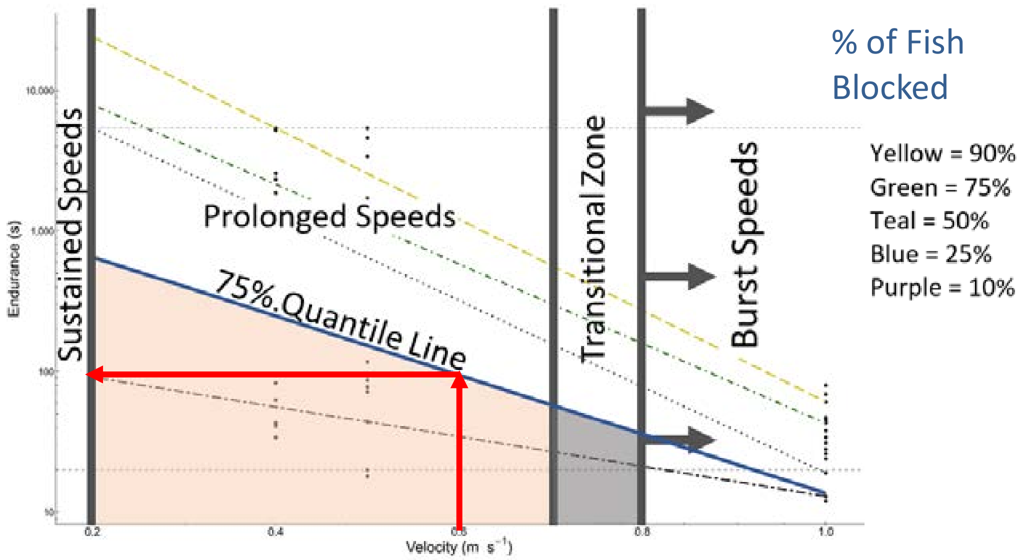
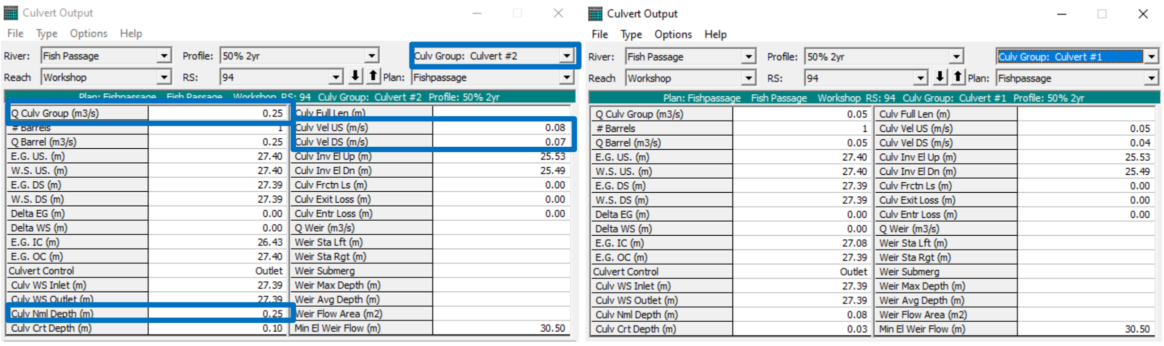


Figure D-2: Preliminary swimming speeds for target species, īnanga.

The culvert output table is a tool in the HEC-RAS 1D model that can be used to obtain detailed information for culvert barrels, as shown in Figure D-3. The total discharge through the primary barrel is 0.25 $m^3 s^{-1}$, with a normal depth of 0.25 m and a maximum velocity of 0.08 $m s^{-1}$ upstream.



Culvert Output

File Type Options Help

River: Fish Passage Profile: 50% 2yr Culv Group: Culvert #2

Reach Workshop RS: 94 Plan: Fishpassage

Plan: Fishpassage Fish Passage Workshop RS: 94 Culv Group: Culvert #2 Profile: 50% 2yr

Q Culv Group (m3/s)	0.25	Culv Full Len (m)	
# Barrels	1	Culv Vel US (m/s)	0.08
Q Barrel (m3/s)	0.25	Culv Vel DS (m/s)	0.07
E.G. US. (m)	27.40	Culv Inv El Up (m)	25.53
W.S. US. (m)	27.40	Culv Inv El Dn (m)	25.49
E.G. DS (m)	27.39	Culv Frctn Ls (m)	0.00
W.S. DS (m)	27.39	Culv Exit Loss (m)	0.00
Delta EG (m)	0.00	Culv Entr Loss (m)	0.00
Delta WS (m)	0.00	Q Weir (m3/s)	
E.G. IC (m)	26.43	Weir Sta Lft (m)	
E.G. OC (m)	27.40	Weir Sta Rgt (m)	
Culvert Control	Outlet	Weir Submerg	
Culv WS Inlet (m)	27.39	Weir Max Depth (m)	
Culv WS Outlet (m)	27.39	Weir Avg Depth (m)	
Culv Nml Depth (m)	0.25	Weir Flow Area (m2)	
Culv Crt Depth (m)	0.10	Min El Weir Flow (m)	30.50

Culvert Output

File Type Options Help

River: Fish Passage Profile: 50% 2yr Culv Group: Culvert #1

Reach Workshop RS: 94 Plan: Fishpassage

Plan: Fishpassage Fish Passage Workshop RS: 94 Culv Group: Culvert #1 Profile: 50% 2yr

Q Culv Group (m3/s)	0.05	Culv Full Len (m)	
# Barrels	1	Culv Vel US (m/s)	0.05
Q Barrel (m3/s)	0.05	Culv Vel DS (m/s)	0.04
E.G. US. (m)	27.40	Culv Inv El Up (m)	25.53
W.S. US. (m)	27.40	Culv Inv El Dn (m)	25.49
E.G. DS (m)	27.39	Culv Frctn Ls (m)	0.00
W.S. DS (m)	27.39	Culv Exit Loss (m)	0.00
Delta EG (m)	0.00	Culv Entr Loss (m)	0.00
Delta WS (m)	0.00	Q Weir (m3/s)	
E.G. IC (m)	27.08	Weir Sta Lft (m)	
E.G. OC (m)	27.39	Weir Sta Rgt (m)	
Culvert Control	Outlet	Weir Submerg	
Culv WS Inlet (m)	27.39	Weir Max Depth (m)	
Culv WS Outlet (m)	27.39	Weir Avg Depth (m)	
Culv Nml Depth (m)	0.08	Weir Flow Area (m2)	
Culv Crt Depth (m)	0.03	Min El Weir Flow (m)	30.50

Figure D-3: HEC-RAS 1D model results, in the primary (left) and secondary (right) barrels.

Step 7. Design complete and ready for review

The design parameters for the culvert are established based on specified design flows: the peak flow (Q_{100}) is calculated at 2.36 $m^3 s^{-1}$, the high Fish Passage Flow (representing 50% of a 2-year event) is 0.3 $m^3 s^{-1}$, and the low Fish Passage Flow (representing 10% of a 2-year event) is 0.06 $m^3 s^{-1}$. Key details of the design include culvert barrels with a diameter of 3,000 mm. The primary barrel is embedded to a depth of 1 m, whereas the secondary barrel is embedded deeper at 1.5 m. The median grain size (D_{50}) of the materials used is 75 mm. It is noted that for the $Q_{50\%\ 2\text{-year}}$, the swimming length exceeds the actual length of the culvert, ensuring adequate passage for fish. Additionally, for the $Q_{10\%\ 2\text{-year}}$, the minimum depth of water within the culvert is maintained at over 150 mm, facilitating safe fish passage. Figure D-4 shows a detailed visual representation of the culvert's dimensions.

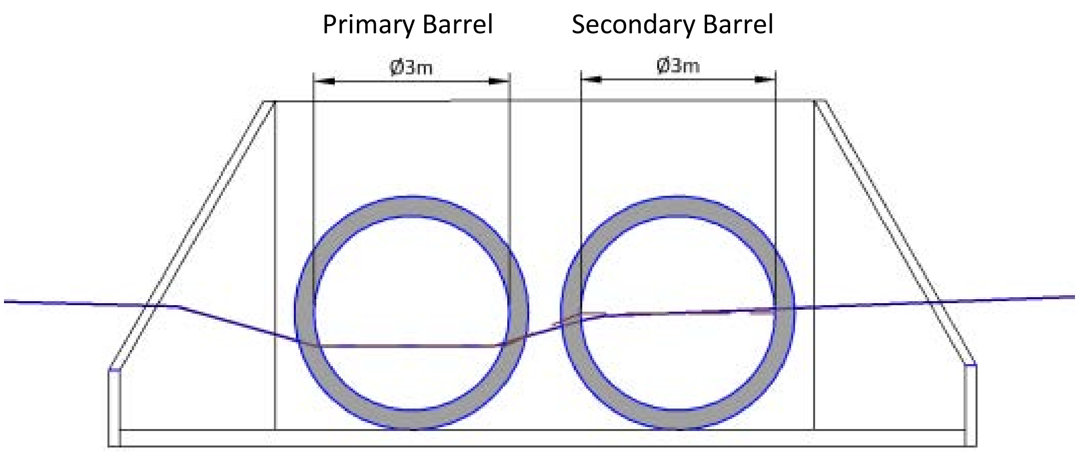


Figure D-4: HEC-RAS 1D model multi-cell round culvert, 3000 mm in diameter.

Appendix E Culvert design worked example 3 (HEC-RAS 1D Version 6.3.1, Multi-Cell Box Culvert Design)

Design procedure application.

The velocity for the fish passage design flow within the culvert is equal to one-third of the culvert's cross-sectional average velocity. The depth for the fish passage design should be based on the depth of 10% of a 2-year flow, which must be greater than 150 mm. Finally, the swimming distance should be more than the culvert length. For round culverts, a minimum embedment is between 1/3 to 1/2 of the culver diameter and for box culverts, the minimum embedment should be either 300 mm or 2 × D_{50}, whichever is greater.

Step 1. Hydrology

Determine design flows

The input parameters required for the HEC-RAS 1D steady model to facilitate fish passage include the fish passage design flow and the downstream normal depth, typically representing the downstream slope. As a part of the design process, the hydrologic characteristics of the sub-catchments that contribute to the stream flow are incorporated into the EPA SWMM 5.1 (SWMM) software for modelling purposes.

The study site has undergone significant modifications, due to urbanisation, therefore bank-full discharge is set to be 50% of 2-year flow as fish passage upper threshold ($Q_{50\%\ 2\text{-year}}$) and 10% of 2-year flow is considered as fish passage lower threshold ($Q_{10\%\ 2\text{-year}}$). Note that, the discharge passing through the primary culvert, in this example, Culvert #2, will be extracted from the HEC-RAS culvert output results Figure E-3. Furthermore, Design Culvert Flow (Q_{100}), corresponds to the flow that is equivalent to a 100-year ARI. The design flows, which have been computed from SWMM model, are summarised in Table E-1.

Table E-1: Design flows used in the HEC-RAS model.

Design Flows Fish Passage Flow	$m^3\ s^{-1}$
Q_{100}	34.66
$Q_{100\text{-year (Culvert \#2)}}$	11.75
$Q_{50\%\ 2\text{-year}}$	3.84
$Q_{50\%\ 2\text{-year (Culvert \#2)}}$	1.90
$Q_{10\%\ 2\text{-year}}$	0.77

Step 2. Stream characteristics

Identify reach characteristics

The minimum extension of the project reach should be either 60 m or three culvert lengths, whichever is greater, from upstream and downstream of the crossing location (HEC-26, 2010). Since the existing culvert is 32 m in length, the project reach must extend at least 96m (three times the culvert lengths) from upstream and downstream of the culvert, Figure E-1.

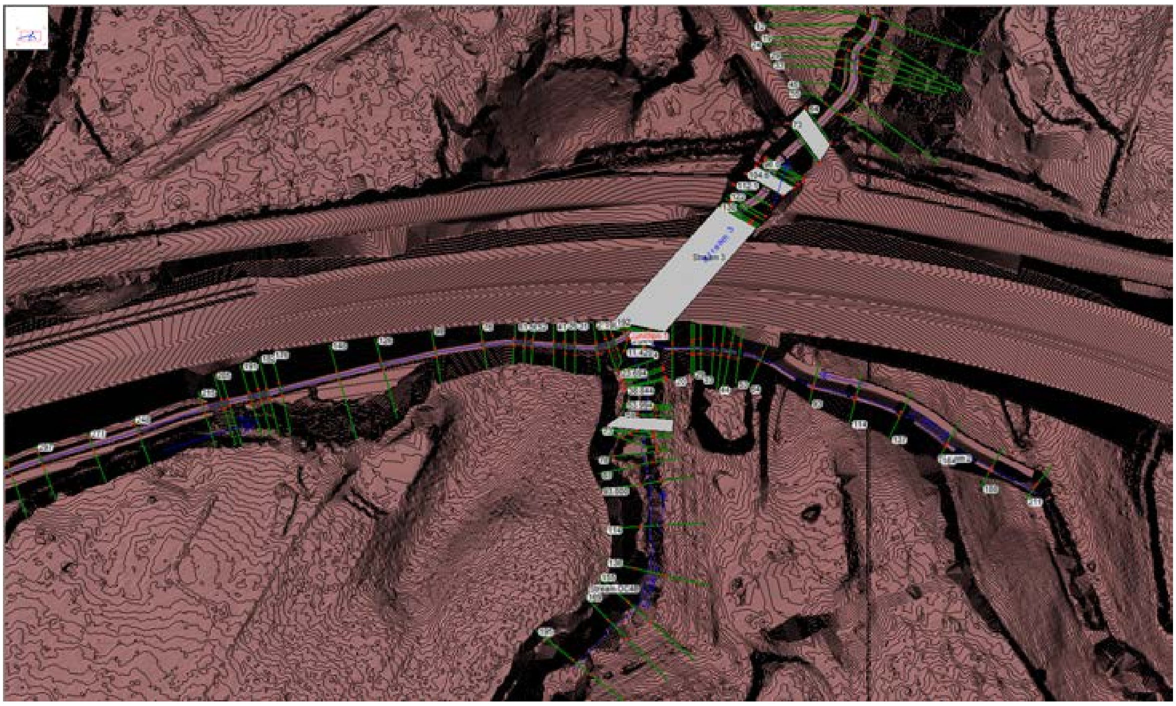


Figure E-1: Site reach and cross-section schematic.

Step 3. Culvert design

Initial culvert design for larger design flows, including the 10-year and 100-year

Culvert diameter shall be selected according to the requirements of the Waka Kotahi NZ Transport Agency or guidelines/standards of the appropriate local council. The culvert configurations, by conducting multiple trials, are designed as a multi-cell box culvert with a main cell with a span of 6 m and a rise of 2.8 m, along with two secondary cells having a span of 6 m and a rise of 2.5 m. The Manning's roughness coefficients are 0.013 for the top and side walls, and 0.085 for the bottom.

Step 4. Culvert embedment design

Determine correct rock size or grain size and embedment depth

The applied shear stress for the culvert (τ) can be obtained using Equation (20). For the primary Culvert #2, data extracted from the Culvert Output in HEC-RAS 1D, as shown in Figure E-3, assuming 100-year flow produces maximum shear stress, the discharge is 11.75 $m^3 s^{-1}$, includes a depth (y) of 1.9 m, a cross-sectional area (A) of 11.4 m^2 (computed as 1.9 m multiplied by 6), and a hydraulic radius (R) of 1.16 m (calculated as 11.4, cross sectional are divided by the, wetted perimeter, sum of 6, width, and twice 1.9 m, depth). The manning roughness is calculated using manning roughness for multiple materials as 0.0628 from Equation (6). From Manning`s equation, the friction slope is:

$$S_f = \left(\frac{Qn}{AR^{2/3}}\right)^2 = \left(\frac{11.75\times0.0628}{11.4\times1.16^{2/3}}\right)^2 = 3.43 \times 10^{-3} \text{ m m}^{-1}$$

Then:

$$\tau = 9810 \times 1.9 \times 3.43 \times 10^{-3} = 63.88 \text{ Pa}$$

The applied shear stress in the culvert is 63.88 Pa, according to Table 4-2, the embedded material required for the culvert is fine cobble-coarse cobble with particle diameters of 150 mm.

In the context of box culverts, the minimum required embedment depth is determined by selecting the greater value between 300 mm or 2 times the D_{50} value. In this example, the embedment depth is designed at 300 mm, as 2 times the D_{50} is 300 mm.

Steps 5 and 6. Test for fish passage

Swimming velocity threshold and water depth

Taking into account that 25% of fish are unable to successfully navigate through the culvert, the endurance duration (s) for target species, at for example 0.6 m s^{-1} swimming speed, can be extracted from Figure E-2.

For Culvert #2, during half of the 2-year, the calculated average velocity at the inlet cross-section is 0.56 m s^{-1}. The adjusted velocity, corresponding to one-third of this measured velocity, is determined to be 0.18 m s^{-1}. From the data shown in Figure E-2, the swimming duration (t_p) for the targeted fish species, īnanga, is 95 seconds. Utilising a swimming velocity of 0.6 m s^{-1} for īnanga, the effective velocity (V_{ef}) is calculated as 0.42 m s^{-1}. This results in a calculated swimming distance of 40 m (0.42 m s^{-1}) × 95 s, which exceeds the culvert length of 32 m. Thus, it is concluded that the targeted species can successfully navigate a distance greater than the culvert's length, indicating adequate passage through Culvert #2. Additionally, the culvert depth for 1/10 of the 2-year is calculated to be 250 mm, which exceeds the minimum required depth, 150 mm, ensuring suitable conditions for fish passage.

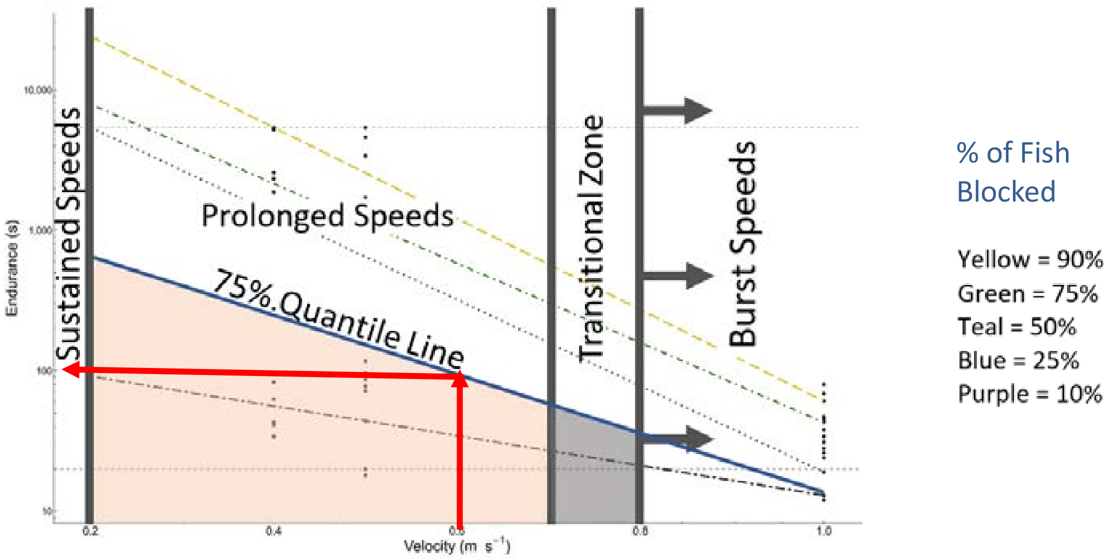
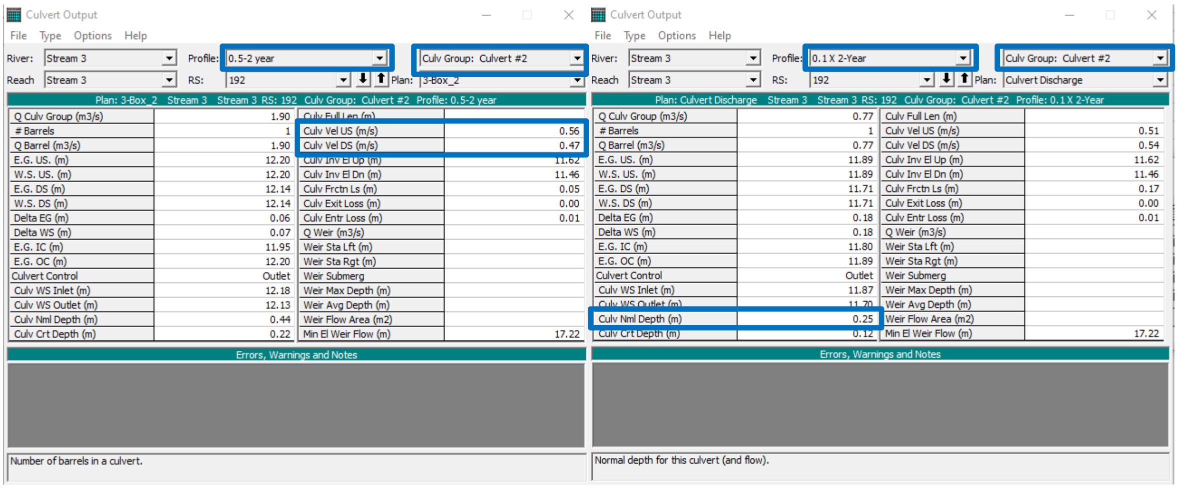


Figure E-2: Preliminary swimming speeds for target species, īnanga.

The culvert output table is a tool in the HEC-RAS 1D model that can be used to obtain detailed information for culvert barrels, as shown in Figure E-3. The total discharge through the primary barrel is 1.9 m^3 s^{-1}, with a normal depth of 0.44 m and a maximum velocity of 0.56 m s^{-1} upstream. The normal depth calculated as 0.25 m in 10% of 2-year event.



Culvert Output

River: Stream 3 | Profile: 0.5-2 year | Culv Group: Culvert #2
Reach: Stream 3 | RS: 192 | Plan: 3-Box_2

Plan: 3-Box_2 Stream 3 Stream 3 RS: 192 Culv Group: Culvert #2 Profile: 0.5-2 year

Q Culv Group (m3/s)	1.90	Culv Full Len (m)	
# Barrels	1	Culv Vel US (m/s)	0.56
Q Barrel (m3/s)	1.90	Culv Vel DS (m/s)	0.47
E.G. US. (m)	12.20	Culv Inv El Up (m)	11.62
W.S. US. (m)	12.20	Culv Inv El Dn (m)	11.46
E.G. DS (m)	12.14	Culv Frctn Ls (m)	0.05
W.S. DS (m)	12.14	Culv Exit Loss (m)	0.00
Delta EG (m)	0.06	Culv Entr Loss (m)	0.01
Delta WS (m)	0.07	Q Weir (m3/s)	
E.G. IC (m)	11.95	Weir Sta Lft (m)	
E.G. OC (m)	12.20	Weir Sta Rgt (m)	
Culvert Control	Outlet	Weir Submerg	
Culv WS Inlet (m)	12.18	Weir Max Depth (m)	
Culv WS Outlet (m)	12.13	Weir Avg Depth (m)	
Culv Nml Depth (m)	0.44	Weir Flow Area (m2)	
Culv Crt Depth (m)	0.22	Min El Weir Flow (m)	17.22

Errors, Warnings and Notes

Number of barrels in a culvert.

Culvert Output

River: Stream 3 | Profile: 0.1 X 2-Year | Culv Group: Culvert #2
Reach: Stream 3 | RS: 192 | Plan: Culvert Discharge

Plan: Culvert Discharge Stream 3 Stream 3 RS: 192 Culv Group: Culvert #2 Profile: 0.1 X 2-Year

Q Culv Group (m3/s)	0.77	Culv Full Len (m)	
# Barrels	1	Culv Vel US (m/s)	0.51
Q Barrel (m3/s)	0.77	Culv Vel DS (m/s)	0.54
E.G. US. (m)	11.89	Culv Inv El Up (m)	11.62
W.S. US. (m)	11.89	Culv Inv El Dn (m)	11.46
E.G. DS (m)	11.71	Culv Frctn Ls (m)	0.17
W.S. DS (m)	11.71	Culv Exit Loss (m)	0.00
Delta EG (m)	0.18	Culv Entr Loss (m)	0.01
Delta WS (m)	0.18	Q Weir (m3/s)	
E.G. IC (m)	11.80	Weir Sta Lft (m)	
E.G. OC (m)	11.89	Weir Sta Rgt (m)	
Culvert Control	Outlet	Weir Submerg	
Culv WS Inlet (m)	11.87	Weir Max Depth (m)	
Culv WS Outlet (m)	11.70	Weir Avg Depth (m)	
Culv Nml Depth (m)	0.25	Weir Flow Area (m2)	
Culv Crt Depth (m)	0.12	Min El Weir Flow (m)	17.22

Errors, Warnings and Notes

Normal depth for this culvert (and flow).

Figure E-3: HEC-RAS 1D model results, culvert output table, in the primary barrel, Culvert #2, at discharges 50% and 10% of 2-Year ARI.

Step 7. Design complete and ready for review

The design parameters for the culvert are established based on specified design flows, with the peak flow (Q_{100}) being 34.66 m³ s^{-1}, the high Fish Passage Flow ($Q_{50\%\ 2\text{-year}}$) at 3.84 m³ s^{-1}, and a high Fish Passage Flow through the primary culvert (Culvert#2) for $Q_{50\%\ 2\text{-year}}$ at 1.90 m³ s^{-1}. The low Fish Passage Flow ($Q_{10\%\ 2\text{-year}}$) is set at 0.77 m³ s^{-1}. Key design details include the main culvert cell featuring dimensions of a 6 m span and a 2.8 m rise, in addition to two secondary cells, each with a span of 6 m and a rise of 2.5 m. All culvert cells are embedded with fine cobble-coarse cobble to a depth of 300 mm. For the $Q_{50\%\ 2\text{-year}}$, the calculated swimming length exceeds the actual length of the culvert, ensuring passage. Furthermore, for the $Q_{10\%\ 2\text{-year}}$ flow, the minimum depth of water within the culvert exceeds 150 mm, facilitating safe fish passage. Figure E-4 shows a detailed visual representation of the culvert's dimensions.

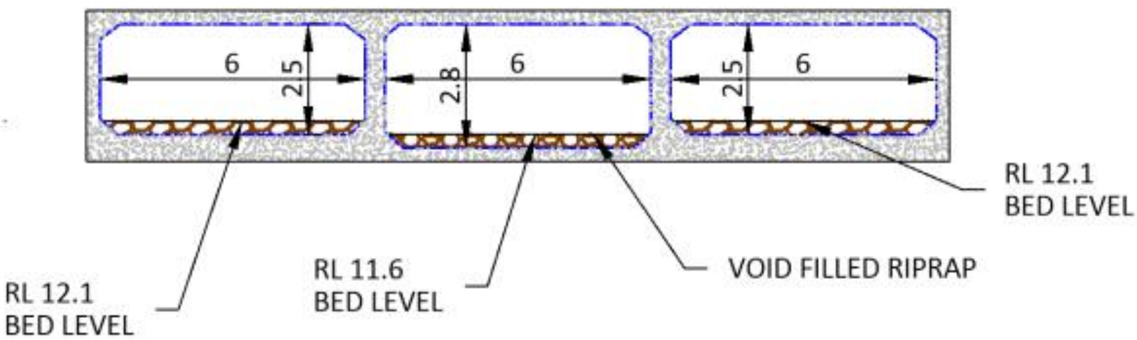


Figure E-4: Multi-cell box culvert on three streams .

Appendix F Culvert design worked example 4 (HEC-RAS 1D Version 6.3.1, Bottomless Culvert Design)

Design procedure application.

The velocity for the fish passage design flow within the culvert is the average cross-sectional velocity from 150 mm away culvert side walls. The depth for the fish passage design should be based on the depth of 10% of a 2-year flow, which must be greater than 150 mm. Finally, the swimming distance should be more than the culvert length. For round culverts, a minimum embedment is between 1/3 to 1/2 of the culver diameter and for box culverts, the minimum embedment should be either 300 mm or 2 × D_{50}, whichever is greater.

Step 1. Hydrology

Determine design flows

The input parameters required for the HEC-RAS 1D steady model to facilitate fish passage include the fish passage design flow and the downstream normal depth, typically representing the downstream slope. As a part of the design process, the hydrologic characteristics of the sub-catchments that contribute to the stream flow are incorporated into the EPA SWMM 5.1 (SWMM) software for modelling purposes.

The study site has undergone significant modification due to urbanisation. As a result, determining the bank-full flow is not appropriate. Therefore, high Fish Passage Flow, $Q_{50\%\ 2\text{-year}}$ is determined for the fish passage at 50% of 2-year, while Low Fish Passage Flow, $Q_{10\%\ 2\text{-year},}$ corresponds to the flow equivalent to 10% of the 2-year ARI flow and Design Culvert Flow, Q_{100} corresponds to the flow equivalent to 100 years. Design Fish Passage Flows computed by SWMM model are summarised in Table F-1.

Table F-1: Design flows used in the HEC-RAS model .

Design Flows	$m^3\ s^{-1}$
Q_{100}	34.66
$Q_{50\%\ 2\text{-year}}$	3.84
$Q_{10\%\ 2\text{-year}}$	0.77

Step 2. Stream characteristics

Identify reach characteristics

The minimum extension of the project reach should be either 60 m or three culvert lengths, whichever is greater, from upstream and downstream of the crossing location (HEC-26, 2010). Since the existing culvert is 32 m in length, the project reach must extend at least 96 m (three times the culvert lengths) from upstream and downstream of the culvert inlet and outlet, Figure F-1.

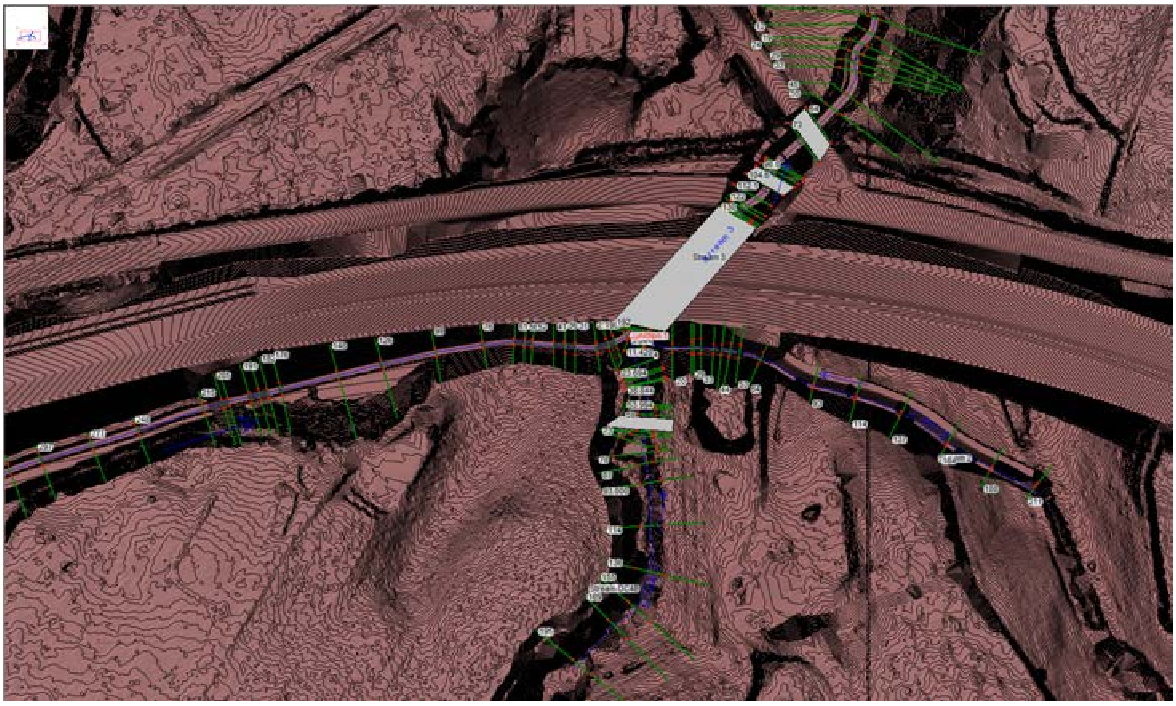


Figure F-1: **Site reach and cross-section schematic.**

Step 3. Culvert design

Initial culvert design for larger design flows, including the 10-year and 100-year

Culvert diameter shall be selected according to the requirements of the Waka Kotahi NZ Transport Agency or guidelines/standards of the appropriate local council. The culvert configurations by conducting multiple trials are designed as a bottomless culvert of 16 m wide. The Manning's roughness coefficients are 0.013 for the top and side walls, and 0.085 for the bottom.

Step 4. Culvert embedment design

Determine correct rock size or grain size and embedment depth

Due to the scour patterns at the entrance of the bottomless culverts, embedment design will vary between maximum applied shear stress for the channel bed and the equation developed for a rip-rap size that accounts for the local velocity at the corner of the culvert entrance by Kerenyi and Pagan-Ortiz (2007) as shown below:

$$D_{50} = \frac{K_r y_0}{(G_s - 1)}\left(\frac{{V_A}^2}{g y_0}\right)^{0.33} \tag{21}$$

Where:

D_{50}	= rip-rap median size (50% finer) (m)
K_r	= sizing Coefficient equal to 0.38 from the best fit lab data, 0.68 for design curve that envelops the lab data
V_A	= average velocity at the culvert entrance (m s^{-1})
y_0	= average flow depth at the culvert entrance before scour (m)

G_s = rip-rap specific gravity

Then:

$$D_{50} = \frac{0.68 \times 0.91}{(2.65 - 1)} \left(\frac{0.41^2}{9.81 \times 0.91} \right)^{0.33} = 0.1\, m = 100\, mm \tag{22}$$

Still the method of shear stress is applied for the bed material within the culvert. To assess the shear stresses exerted on the bed material within the culvert, hydraulic analyses are conducted. The applied shear stress for the culvert (τ) can be obtained using Equation (20)

For the bottomless culvert, data extracted from the Bridge Output in HEC-RAS 1D includes a depth (y) of 2.27 m, a cross-sectional area (A) of 33.88 m^2 (extracted from total cross sectional areas), and a hydraulic radius (R) of 1.02 m (calculated as 33.88, cross sectional are divided by the, wetted perimeter, 33.14). The manning roughness is calculated using manning roughness for multiple materials as 0.0585 from Equation (6). From Manning`s equation, the friction slope is:

S_f $= (\frac{Qn}{AR^{2/3}})^2 = (\frac{34.66 \times 0.0585}{33.88 \times 1.02^{2/3}})^2 = 3.47 \times 10^{-3}$ m m^{-1}

Then:

$$\tau = 9810 \times 2.27 \times 3.47 \times 10^{-3} = 77 \text{ Pa}$$

The applied shear stress in the culvert is 77 Pa, according to Table 4-2 the embedded material required for the culvert is fine-coarse cobble, with D_{50} of 180 mm. In terms of bottomless culverts, the minimum depth of embedment is determined by selecting the greater value between 300 mm or 2 times the D_{50} value. In this example, the embedment depth is set at 360 mm.

Steps 5 and 6. Test for fish passage

Swimming velocity threshold and water depth

Taking into account that 25% of fish are unable to successfully navigate through the culvert, the endurance duration (s) for target species, at for example 0.6 m s^{-1} swimming speed, can be extracted from Figure F-2.

For half of the 2-year, the calculated average velocities near the culvert wall—within 0.15 m of culvert walls—are 0.23 m s^{-1} upstream (as shown in Figure F-3) and 0.20 m s^{-1} downstream (as shown in Figure F-4). Based on the swimming duration data from Figure F-2, which indicates a duration (t_p) of 95 seconds and using the swimming velocity of the targeted fish species, īnanga, at 0.6 m s^{-1} while considering the highest velocity, the effective velocity (V_{ef}) is calculated to be 0.37 m s^{-1}. This results in a swimming distance of 35 m ((0.37 m s^{-1}) × 95 s), which exceeds the culvert length of 32 m. Thus, it is concluded that īnanga can successfully navigate a distance greater than the culvert's length, indicating that the target fish species can adequately pass through the culvert. Additionally, the culvert depth for 1/10 of the 2-year is calculated to be 0.25 m, which exceeds the minimum depth requirement of 0.15 m, as evidenced in Figure F-4, ensuring suitable conditions for fish passage.

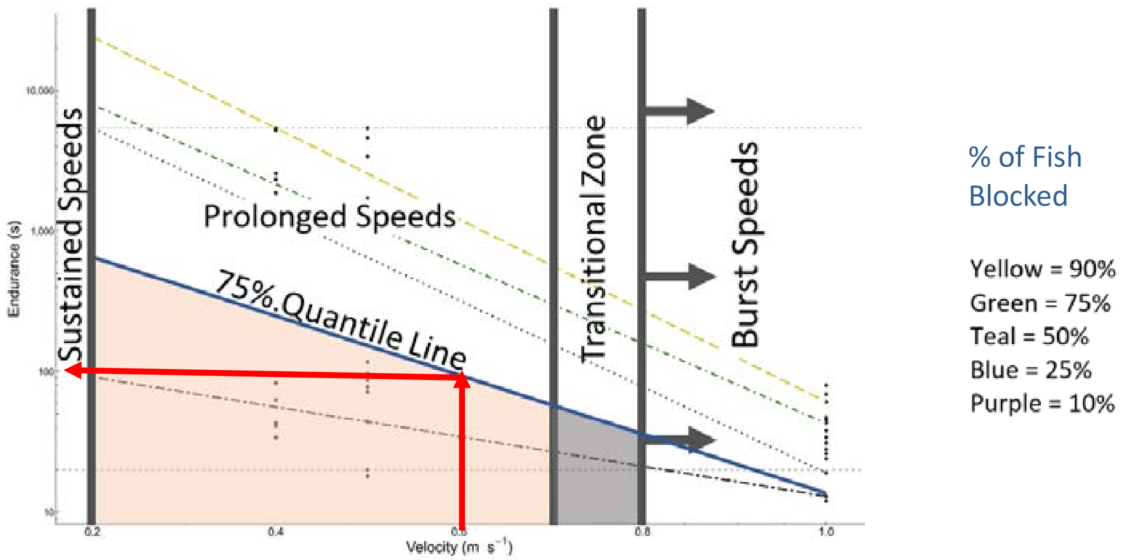


Figure F-2: Preliminary swimming speeds for target species, īnanga.

The culvert Flow Distribution Output table is a tool in the HEC-RAS 1D model that can be used to obtain detailed information for culvert, as shown in Figure F-3 and Figure F-4. The average cross-sectional velocity approximately 0.15 m from the culvert side walls for 50% of a 2 year is calculated as 0.23 m s^{-1} and 0.20 m s^{-1}, at the upstream and downstream, respectively.

Flow Distribution Output

File Type Options Help

River: Stream 3 Profile: 0.5-2 year

Reach Stream 3 RS: 192 BR U Plan: Plan 18

Plan: Plan 19 Stream 3 Stream 3 RS: 192 BR U Profile: 0.5-2 year

	Pos	Left Sta (m)	Right Sta (m)	Flow (m3/s)	Area (m2)	W.P. (m)	Percent Conv	Hydr Depth(m)	Velocity (m/s)	Shear (N/m2)	Power (N/m s)
1	LOB	2.21	5.01	0.01	0.03	0.36	0.19	0.21	0.23	3.56	0.81
2	Chan	5.01	8.58	0.30	1.11	3.57	7.69	0.31	0.27	12.42	3.29
3	Chan	8.58	12.15	0.86	2.13	3.65	22.41	0.60	0.40	23.31	9.40
4	Chan	12.15	15.72	1.76	3.25	3.58	45.83	0.91	0.54	36.23	19.61
5	Chan	15.72	19.29	0.66	1.81	3.64	17.07	0.51	0.36	19.82	7.18
6	Chan	19.29	22.86	0.26	1.06	3.78	6.81	0.30	0.25	11.16	2.75

Figure F-3: Upstream flow distribution output for 1/2 of 2-year ARI design flow. Average velocity is 0.23 m s^{-1} in a cross section 0.15 m from the culvert wall.

Flow Distribution Output

File Type Options Help

River: Stream 3 Profile: 0.5-2 year

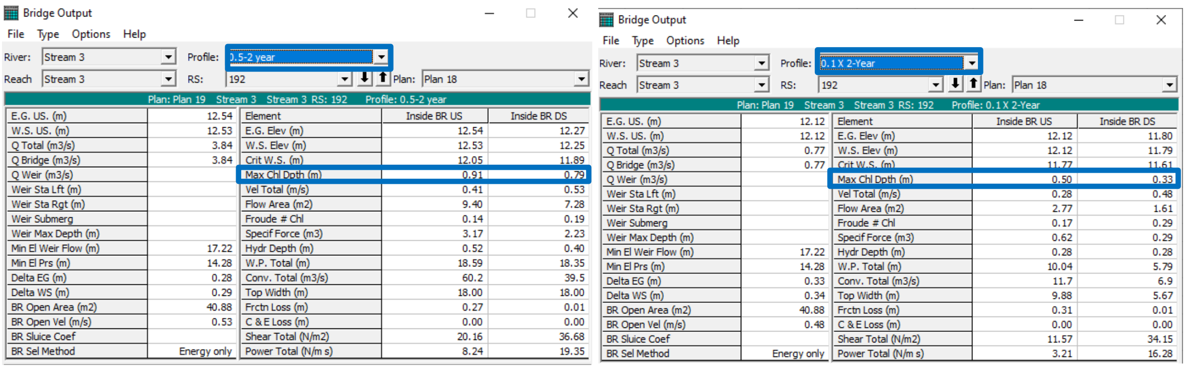
Reach Stream 3 RS: 192 BR D Plan: Plan 18

Plan: Plan 19 Stream 3 Stream 3 RS: 192 BR D Profile: 0.5-2 year

	Pos	Left Sta (m)	Right Sta (m)	Flow (m3/s)	Area (m2)	W.P. (m)	Percent Conv	Hydr Depth(m)	Velocity (m/s)	Shear (N/m2)	Power (N/m s)
1	LOB	0.12	0.90	0.00	0.01	0.24	0.08	0.10	0.20	5.53	1.13
2	Chan	0.90	4.47	0.19	0.68	3.57	4.85	0.19	0.27	17.55	4.82
3	Chan	4.47	8.04	0.76	1.59	3.65	19.80	0.45	0.48	40.30	19.27
4	Chan	8.04	11.61	2.02	2.83	3.57	52.61	0.79	0.71	73.41	52.35
5	Chan	11.61	15.18	0.70	1.51	3.65	18.20	0.42	0.46	38.32	17.72
6	Chan	15.18	18.75	0.17	0.65	3.67	4.46	0.18	0.26	16.42	4.32

Figure F-4: Downstream flow distribution output for 1/2 of 2-year ARI design flow. Average velocity is 0.20 m s^{-1} in a cross section 0.15 m from the culvert wall.

The culvert Bridge Output table is a tool in the HEC-RAS 1D model that can be used to obtain detailed information for culvert, as shown in Figure F-5. The cross-sectional average velocity for 50% of a 2 year is calculated as 0.41 m s^{-1} and 0.53 m s^{-1} at the upstream and downstream of the culvert, respectively. The maximum channel depth is also calculated as 0.50 m and 0.33 m, at the upstream and downstream of the culvert for 10% of 2-year flow, respectively, Figure F-5.



Bridge Output
File Type Options Help
River: Stream 3 Profile: 0.5-2 year
Reach Stream 3 RS: 192 Plan: Plan 18

Plan: Plan 19 Stream 3 Stream 3 RS: 192 Profile: 0.5-2 year

E.G. US. (m)	12.54	Element	Inside BR US	Inside BR DS
W.S. US. (m)	12.53	E.G. Elev (m)	12.54	12.27
Q Total (m3/s)	3.84	W.S. Elev (m)	12.53	12.25
Q Bridge (m3/s)	3.84	Crit W.S. (m)	12.05	11.89
Q Weir (m3/s)		Max Chl Dpth (m)	0.91	0.79
Weir Sta Lft (m)		Vel Total (m/s)	0.41	0.53
Weir Sta Rgt (m)		Flow Area (m2)	9.40	7.28
Weir Submerg		Froude # Chl	0.14	0.19
Weir Max Depth (m)		Specif Force (m3)	3.17	2.23
Min El Weir Flow (m)	17.22	Hydr Depth (m)	0.52	0.40
Min El Prs (m)	14.28	W.P. Total (m)	18.59	18.35
Delta EG (m)	0.28	Conv. Total (m3/s)	60.2	39.5
Delta WS (m)	0.29	Top Width (m)	18.00	18.00
BR Open Area (m2)	40.88	Frctn Loss (m)	0.27	0.01
BR Open Vel (m/s)	0.53	C & E Loss (m)	0.00	0.00
BR Sluice Coef		Shear Total (N/m2)	20.16	36.68
BR Sel Method	Energy only	Power Total (N/m s)	8.24	19.35

Bridge Output
File Type Options Help
River: Stream 3 Profile: 0.1 X 2-Year
Reach Stream 3 RS: 192 Plan: Plan 18

Plan: Plan 19 Stream 3 Stream 3 RS: 192 Profile: 0.1 X 2-Year

E.G. US. (m)	12.12	Element	Inside BR US	Inside BR DS
W.S. US. (m)	12.12	E.G. Elev (m)	12.12	11.80
Q Total (m3/s)	0.77	W.S. Elev (m)	12.12	11.79
Q Bridge (m3/s)	0.77	Crit W.S. (m)	11.77	11.61
Q Weir (m3/s)		Max Chl Dpth (m)	0.50	0.33
Weir Sta Lft (m)		Vel Total (m/s)	0.28	0.48
Weir Sta Rgt (m)		Flow Area (m2)	2.77	1.61
Weir Submerg		Froude # Chl	0.17	0.29
Weir Max Depth (m)		Specif Force (m3)	0.62	0.29
Min El Weir Flow (m)	17.22	Hydr Depth (m)	0.28	0.28
Min El Prs (m)	14.28	W.P. Total (m)	10.04	5.79
Delta EG (m)	0.33	Conv. Total (m3/s)	11.7	6.9
Delta WS (m)	0.34	Top Width (m)	9.88	5.67
BR Open Area (m2)	40.88	Frctn Loss (m)	0.31	0.01
BR Open Vel (m/s)	0.48	C & E Loss (m)	0.00	0.00
BR Sluice Coef		Shear Total (N/m2)	11.57	34.15
BR Sel Method	Energy only	Power Total (N/m s)	3.21	16.28

Figure F-5: HEC-RAS 1D model results, bridge output table, in the bottomless culvert, for 50% And 10% of 2-Year ARI.

Step 7. Design complete and ready for review

The design parameters for the culvert are, design culvert flow (Q_{100}) of 34.66 $m^3 s^{-1}$, a High Fish Passage Flow ($Q_{50\%\ 2\text{-year}}$) of 3.84 $m^3 s^{-1}$, and a Low Fish Passage Flow ($Q_{10\%\ 2\text{-year}}$) of 0.77 $m^3 s^{-1}$. Key design features include the culvert being bottomless with a span of 16 m, utilisation of 100 mm rip-rap stone at the culvert's corner inlet, and the primary construction material being fine cobble-coarse cobble with D_{50} of 180 mm and an embedment depth of 360 mm. During the $Q_{50\%\ 2\text{-year}}$, the swimming length exceeds the actual length of the culvert, and during the $Q_{10\%\ 2\text{-year}}$, the minimum water depth remains above 150 mm, ensuring adequate conditions for fish passage. Figure F-6 shows a detailed visual representation of the culvert's dimensions.

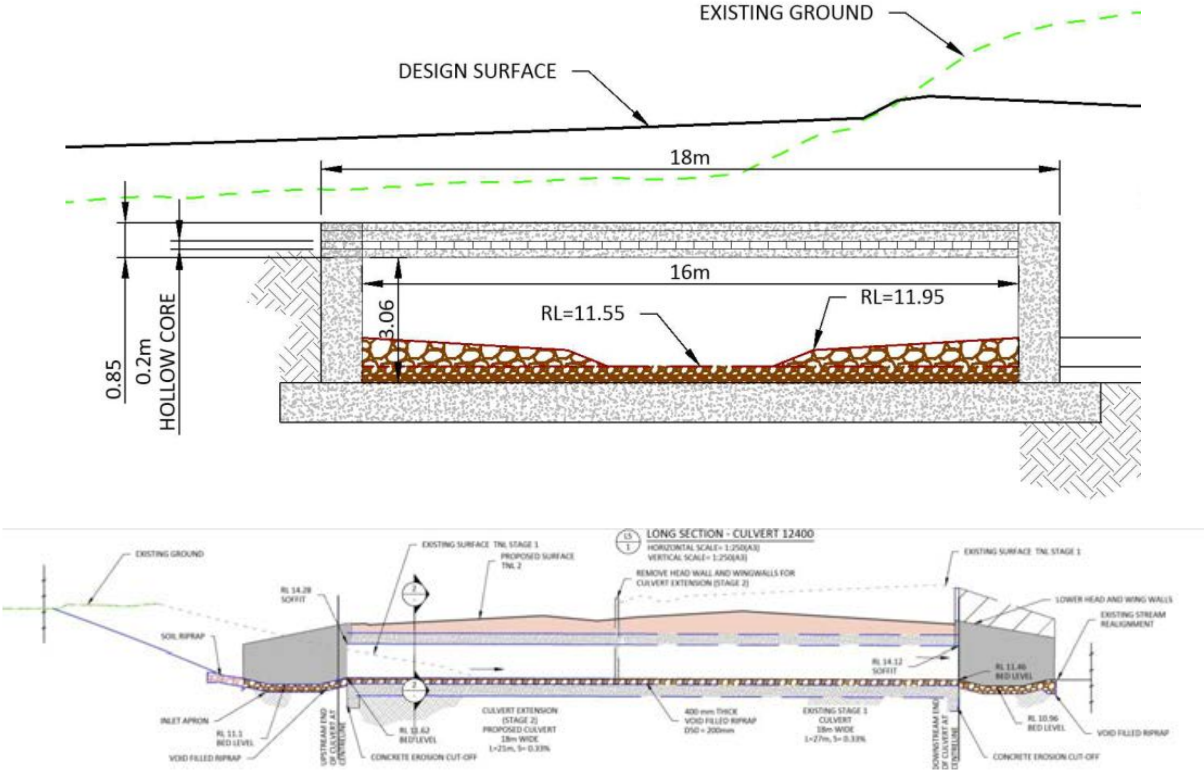


Figure F-6: Bottomless culvert, 16 m wide, and 32 m long.

Appendix G Examples of soil rip-rap application

Figure G-1A shows a portion of a gully relocation, that made way for a large grade separated intersection. The original gully was extremely incised and unstable, with very limited ecological value. The figures show a bend in the stream that was designed with a subtle bend apex pool configuration. This required a steep bank on the outside of the curve, which was reinforced with soil rip-rap. The photo was taken shortly after the stream was livened, which was 6 months after planting.

Figure G-1B shows the same bend in the stream, approximately 2 years later. The stream had experienced several abnormally high flows without any perceivable damage to the stream. Within the bend the planting combined with the soil rip-rap supported the steeper bank slope without any erosion.

A

B

Figure G-1: Examples of soil rip-rap application.

Figure G-2 shows the application of soil rip-rap at an outfall from a severely surcharged reticulated stormwater system during the spring season following its late summer construction in 2010. Note that soil rip-rap was used in the impact basin as well. The sub-catchment that this outfall serves experienced several extreme events during its first few years, but the combination of soil rip-rap and vegetation have withstood the erosive forces of the high velocity outflow. Figure G-3 shows a series of aerial views of the same outfall over four years after construction.

Figure G-2: Application of soil rip-rap at a stormwater outfall.

Figure G-3: Vegetation growth as part of soil rip-rap at stormwater outfall over four years following construction.

Figure G-4 shows another example of the application of soil rip-rap over a four-year period, starting in 2012. This project involved the construction of a stream through a stabilised toxic waste site. A 1-metre soil cap had to be maintained over soils contaminated with several toxic substances, including high concentrations of metals.

Soil rip-rap was used for the channel, the banks, and the overbanks. Three boulder grade control structures are visible in the aerial photos. The channel bottom was planted with wetland species and the overbanks were seeded and mulched and some scattered shrubbery plantings were added.

Due to dry cold winters and hot dry summers, plant growth is very slow. This project has also been subjected to multiple extreme events, including one that met the 100-year design capacity, yet the soil and vegetation remained. The design flood event occurred prior to the 2012 aerial photos.

Both projects are within the City and County of Denver, Colorado, which is ordinarily quite dry with high intensity storms, often leading to flash flooding.

Figure G-4: Vegetation growth as part of soil rip-raprip-rap over four years following construction of a stream through a stabilised toxic waste site.

Appendix H Scour protection specifications

Rock rip-rap scour protection

The rock rip-rap material shall comprise of imported screened quarry blasting and/or crushed basalt, greywacke or other hard rock which shall conform to the following (Richardson et al. 2001; Thompson and Kilgore 2006):

- The rip-rap designation and total thickness of rip-rap shall be as shown on the Drawings. The maximum stone size shall not be larger than the thickness of the rip-rap and neither width nor thickness of a single stone of rip-rap shall be less than one-third of its length.
- Density: Greater than 2.40 t m^3, when tested in accordance with NZS 4407:1991 Test 3.7.2.
- Crushing Resistance: Not less than 150 kN, when tested in accordance with NZS 4407:1991 Test 3.10.
- Weathering Classification: AA, BA, AB or BB when tested in accordance with NZS 4407:1991 Test 3.11.
- Abrasion Resistance: Not more than 26%, when tested in accordance with NZS 4407:1991 Test 3.12.
- Rock shall be free of calcite intrusions.
- Broken concrete or asphalt pavement shall not be acceptable for use in the work.

Rip-rap rock shall be well graded as per Table H-1below.

Table H-1: Rip-rap gradation.

Stone size range (m)	Stone weight range (kg)	% of gradation smaller than
1.5 D_{50} to 1.7 D_{50}	3.0 W_{50} to 5.0 W_{50}	100
1.2 D_{50} to 1.4 D_{50}	2.0 W_{50} to 2.75 W_{50}	85
1.0 D_{50} to 1.4 D_{50}	1.0 W_{50} to 1.5 W_{50}	50
0.4 D_{50} to 0.6 D_{50}	0.1 W_{50} to 0.2 W_{50}	15

No rock shall be placed until the Engineer is satisfied that the material meets the requirements above. A non-woven geotextile shall be provided under all rip-rap with the ends well anchored. The strength and filtration class of the geotextile shall be selected in accordance with the advice given in the NZTA F/7 and accompanying notes.

Rip-rap placement shall commence as follows:

- Rip-rap shall be placed on the prepared slope and/or channel bottom areas in a manner that will produce a reasonably well graded mass of stone.
- Rip-rap shall be machine placed, unless otherwise stipulated in the Drawings or Contract.

- There shall be no large accumulations of either the larger or smaller sizes of stone, and some hand placement may be required to achieve this.
- Unless otherwise authorised by the Engineer, the rip-rap protection shall be placed in conjunction with the construction of embankment or channel bottom.
- All rock is to be placed in a dewatered condition beginning at the toe of the slope or other lowest point, working upwards.
- The entire mass of rip-rap shall be placed in conformance with the gradation table above, to line, grade, and thickness shown on the Drawings.
- Rip-rap shall be placed to full course thickness at one operation and in a manner that avoids displacing the underlying bedding material. Placing of rip-rap by dumping into chutes, or by similar methods shall not be permitted.
- The procedure shall result in larger materials flush to the top surface with faces and shapes arranged to minimise voids, and smaller material below and between larger materials.
- Projections above or depressions under the finished design grade shall not be more than 10% of the rock layer thickness.
- Smaller rocks shall be securely locked between the larger stones.
- The stone shall be consolidated by the bucket of the backhoe or other means that will cause interlocking of the material.

The Contractor is responsible for the maintenance of the rip-rap protection until the Defects Liability Period is over. Any material displaced for any reason shall be replaced to the lines and grades shown on the Drawings at no additional cost to the Principal. If the bedding materials are removed or disturbed, such material shall be replaced prior to replacing the displaced rip-rap.

Soil rip-rap

The rip-rap rock (conforming to 1.32.1) shall be pre-mixed with topsoil from on site at a ratio of 70% rip-rap to 30% soil by volume. Soil rip-rap shall consist of a uniform mixture of soil and rip-rap without voids. Geotextile or granular bedding are not necessary under soil rip-rap, except where specifically shown on the drawings. Geotextile or granular bedding are not required where soil rip-rap is specified.

The above rip-rap placement methodology (1.32.1) shall be modified as follows:

- Adjacent stockpiles of rip-rap and soil shall be created near the placement site and mixed prior to placement.
- Mix 30% soil by volume with stockpiled rip-rap, using additional moisture and control procedures that ensure a homogenous mixture, so that the soil fills the inherent voids in the rip-rap without displacing rip-rap.
- Place a first layer of smaller soil rip-rap of approximate D_{50} thickness. Then place the top layer with surface rocks that are largely D_{50} or greater, filling voids as necessary with smaller rip-rap.

- The mixture shall be consolidated by large vibratory equipment or backhoe bucket to create a tight, dense interlocking mass.
- The soil shall be further wetted to encourage void filling with soil.
- Any large voids shall be filled with rock and small voids filled with soil.
- Excessively thick zones of soil ($0.3D_{50}$) prone to washing away shall not be created (for example, no thicknesses greater than 150 mm).
- For buried soil rip-rap, the top surface shall be covered with 100 mm of topsoil such that no rock points are protruding.
- The final surface shall be thoroughly wetted for good compaction, smoothed, and compacted by vibrating equipment; the surface shall then be hand raked to receive planting or seeding.

Void-filled rip-rap

Where void-filled rip-rap is specified on the Drawings, the rip-rap rock shall be premixed with well graded granular material with a maximum particle size of 40 mm (GAP 40) at a ratio of 70% rip-rap to 30% GAP 40 by volume.

The void filler is a mixture of graded and passing sand, gravel, and rock. Gradation of the void filler varies in relation to the size (D_{50}) of the rip-rap. Table H-2 and Table H-3 provide mix proportions for rip-rap sizes from 150 mm D_{50} to 450 mm D_{50}.

Table H-2: Mix requirements for D_{50} 150 mm to D_{50} 250 mm void filled rip-rap.

Approximate Proportions (Loader or Excavator Buckets)	Material Type	Material Description
6	Rip-rap	150 mm D_{50} to 250 mm D_{50}
3	Void Fill (granular)	Clean GAP 100
½ to 1	Void Fill	Native or in-situ topsoil (topsoil not to be added for work in live streams with medium to coarse sand or gravel/cobble beds)

Table H-3: Mix requirements for D_{50} 300 mm to D_{50} 450 mm void filled rip-rap.

Approximate Proportions (Loader or Excavator Buckets)	Material Type	Material Description
6	Rip-rap	300 mm D_{50} to 450 mm D_{50}
1	Void Fill (Small Rock)	Clean Rock, well graded from 200 mm to 100 mm
2	Void Fill (granular)	Clean GAP 100

Mix proportions within Table H-2 and Table H-3 are subject to adjustment by the Engineer. Where required rip-rap sizes exceed 450 mm D_{50}, bespoke rip-rap and void fill must be provided. Monitoring and mitigation of void fill loss, resulting in more than 30% exposure of the top layer of rip-rap is required. Geotextile or granular bedding are not necessary under void-filled rip-rap except where specifically shown on the drawings.

The above rip-rap placement methodology shall be modified as follows:

- Rip-rap and void filler shall be mixed before placement in the proportions shown in Table H-2 and Table H-3.
- Place a first layer of smaller void fill rip-rap of approximate D_{50} thickness. Then place the top layer with surface rocks that are largely D_{50} or greater, filling voids as necessary with smaller rip-rap.
- The mixture shall be consolidated by large vibratory equipment or backhoe bucket to create a tight, dense interlocking mass.
- The mixture shall be further wetted to encourage void filling with fines.
- Any large voids shall be filled with rock and void filler, as appropriate.
- Excessively thick zones of void filler shall not be created.
- The final surface shall be thoroughly wetted for good compaction, smoothed, and compacted by vibrating equipment.
- Where planting is indicated, the surface shall be hand covered with 150 mm (min) of topsoil and raked to receive planting or seeding.

Boulders and grouted boulders

- Boulders shall have a specific gravity greater than 2.1 and be clean and free of cracking. Boulders must be approved by the Engineer, prior to placement.
- The minimum dimension of each boulder must be the size indicated in the drawings.
- The maximum dimension of each boulder shall not exceed 1.5 × the minimum dimension indicated in the drawings.
- Grout used for grouted boulder structures shall meet the requirements.
- Boulders are to be placed in a manner that meets the lines and grades provided on the drawings. The number of boulders shown in the drawings is indicative.
- The subgrade to receive boulders shall be excavated and any unstable material shall be removed and replaced to achieve a stable subgrade. Where plans call for placed bedding material, the bedding will be free of loose soil or other debris or detritus.
- Subgrade shall be excavated a minimum of 150 mm and a maximum of 300 mm behind the boulders.
- Keep grout contact surfaces on boulders always wet prior to receiving grout.

- Smaller rocks may be ‘chinked in’ to fill voids behind the boulders. Smaller rocks shall also be used to reduce gaps larger than four 100 mm. Placement shall be approved by an engineer prior to grouting.
- The top of the boulder structure shall be at the level indicated on the plans. The top level of individual boulders, in the top of the structure, shall be no more than 50 mm different than the level indicated on the Drawings.
- Contractor shall inform the engineer at least 24 hours in advance of grout placement, to allow inspection of the boulder placement. If the contractor places the grout without prior engineer approval of boulder placement and the boulder placement is not accepted by the engineer, the contractor will be required to remove the improperly placed grouted boulders and replace them with clean boulders, placed properly and re-grouted after approval of the placement.
- A vibrator small enough to fit between the boulders (35 mm) shall be used to make sure all voids are filled between the boulders from the subgrade and around the boulders to a depth as shown on the drawings. Contractor shall use a wood float or brush, when approved by engineer, to smooth and grade the grout around the boulders.
- Grout between boulders shall be recessed ⅓ the diameter of the boulders on the side facing the channel.
- Grout should be raked out and finished to minimise visibility.
- Clean and wash any spillage before the grout sets so the visual surfaces of boulders will be free of grout to provide a clean, natural appearance, or if washing does not clean off grout residue, contractor shall wash off any grout residue with a wire brush or muriatic acid and water, using a brush to scrub off the residue.
- Grout shall receive cold or hot weather protection in accordance with Section 03 31 00, Structural Concrete.
- The Contractor shall complete grouted boulder structures in a single pour as often as practicable. When large structures require separate pours, Cold Joints shall meet the following requirements:
- No Cold Joints shall be allowed within grouted boulder basins.
- A 1.0 m piece of 6 mm rebar shall be installed as a starter between each boulder at the end of the initial pour. The bars will be embedded 500 mm into the concrete at the end of the initial pour.
- Cold Joints shall have a roughened edge and be clean and free of dirt or other debris.
- Cold Joints in grouted boulder structures shall have a non-linear “jigsaw” shaped edge.
- Cold Joints on cut-off walls must have a keyway formed in the top of the initial pour. The keyway shall be a minimum of 50 mm deep and 100 mm wide. 6 mm rebar, 600 mm long shall be embedded 300 mm, at 300 mm centres, into the keyway of the initial pour.

Appendix I Remediation evidence synthesis: Part 1 – artificial ramps

Overview

In New Zealand, a range of artificial substrate ramps have been tested as the basis of designing a cost-effective solution for overcoming low-head vertical drops, for example, downstream of perched culverts. To underpin the recommendations made in these guidelines we synthesised current evidence on key ramp design features for New Zealand species.

Methodology

We collated evidence from the peer-reviewed literature and grey literature resources evaluating the performance of different ramp designs for facilitating the upstream movement of small-bodied fishes in New Zealand. The evidence synthesis was restricted to evaluating the evidence for artificial ramp designs.

Greatest weight was placed on evidence from peer-reviewed literature, acknowledging the rigorous review process these sources are subject to.

Results

Artificial substrate ramps

Based on the results of studies by Baker and Boubée (2006), Doehring et al. (2012), Baker (2014), Jellyman et al. (2016), Fake (2018), and Franklin et al. (2021), there is clear evidence that ramp substrate, length and slope, and the provision of wetted margins, are all important considerations in artificial ramp designs.

High passage success (≥90%) was limited to ramps with a roughened substrate that were ≤1.5 m long with a slope of 15° (Baker and Boubée 2006). The Miradrain™ substrate provided the best results across the range of species and substrates that have been tested (Baker and Boubée 2006). Doubling the length of a 15° Miradrain™ ramp from 1.5 m to 3 m reduced passage success by approximately 30 and 50% for īnanga and redfin bully respectively (Baker 2014). Common bully passage success over the 3 m, 15° Miradrain™ ramp was <15%. This indicates that at slopes of up to 15°, ramp length for a Miradrain™ type ramp should generally be limited to ≤1.5 m to optimise passage success. Addition of spat ropes to Miradrain™ ramps made no statistically significant difference to overall passage success for īnanga and redfin bully, but may provide some benefit for the passage of small individuals (Fake 2018). At a slope of 2–3° on a ramp with Miradrain™, īnanga were able to successfully travel up to 7 m (maximum length tested) (Hicks et al. 2008). Trials with non-Miradrain™ substrates at lower ramp gradients showed passage rates for īnanga approximately doubled over a 3m distance with a reduction in ramp slope from 15° to 5° (Doehring et al. 2012).

A rotational moulded plastic ramp designed to mimic Miradrain™ has been developed. The ramps are 560 mm wide and 2.4 m long and can be cut to length on site. They have a V-shaped cross-sectional profile and include baffles that mimic the size and configuration of the Miradrain™ substrate. The ramps offer a cost-effective implementation of the artificial ramp design that can be retrospectively installed at the downstream end of culverts or weirs using flexible attachments to provide short-term remediation. Testing of these ramps has indicated passage rates for īnanga equivalent to previous ramp experiments, but lower success rates for redfin bully (Fake 2018). Initial field testing has demonstrated that the moulded plastic ramps are robust in-situ in the short-term but can be subject to wear and tear that will impair their effectiveness following longer-term (>2 years) deployment. The authors have recently observed substrates similar to Miradrain™ that have been installed on artificial ramps in the field. It is important to note that other cupped substrates have either not been laboratory tested or certain shapes (e.g., smaller cups) were deemed unsuitable when test substrates were determined by Baker and Boubée (2006).

Another variation on the artificial ramp design is the floating ramp. A key point of difference for the floating artificial ramps compared to other designs that have been tested is that the downstream end of the ramp floats rather than being fixed to the river/stream bed. It is hypothesised that difficulties in finding the floating ramp entrance are likely the cause of the lower success rates observed for redfin bully on these ramps compared to other studies. Redfin bully are a benthic species that inhabit the stream bed and so are less likely to encounter or be attracted to a ramp entrance that does not extend to the river bed. Consequently, we recommend caution in using floating ramps where fish passage objectives include providing passage for benthic species. Addition of spat rope to these floating ramps did improve passage success for redfin bully (but not for other species tested), even though it made no difference on the Miradrain™ ramps this design mimics (Fake 2018). The reason for this difference in performance is unclear and requires further testing. Given the results of previous ramp studies (e.g. Baker and Boubée 2006; Baker 2014), fixing the downstream end of these ramps so that they reach the stream bed may offer improved passage for benthic species, but this has not yet been evaluated. Exact replication of the Miradrain™ substrate dimensions on these ramps is expected to be important for ensuring effective performance given the relatively poorer performance of similar substrates with different dimensions (e.g., cup size) in initial testing (Baker and Boubée 2006).

Flexible (or rubber) ramps

'Flexible ramps' or 'Rubber ramps' have increasingly been deployed as a fish passage mitigation method at perched culverts in New Zealand (e.g., Figure I-1; Olley et al. (2024)). The evidence base supporting the effectiveness of these ramps remains weak. Multiple studies have demonstrated that passage efficiency on smooth substrates is poor to zero for climbing and non-climbing native fish species (e.g. Baker and Boubée 2006; Jellyman et al. 2016). Mark-recapture data from one site with a rubber ramp installed with spat ropes showed <1% passage success for īnanga over a 48hr period (Baker et al. 2024b). Similarly, Olley et al. (2024) showed that īnanga were unable to pass rubber ramps with spat ropes at multiple sites.

Where evidence of increased passage following installation of rubber ramps has been presented (e.g., Olley et al. (2024) showed increased passage of banded kōkopu), mussel spat ropes have also been installed concurrently. The presence of both the rubber ramp and spat rope confounds any conclusion regarding the benefits of the ramp itself as improvements in the passage of climbing species such as banded kōkopu following the installation of spat ropes at a perched culvert is consistent with previous published work (David and Hamer 2012). See Section 5.5.7 for more details on the use of spat ropes for fish passage remediation. There is some anecdotal evidence of climbing fish species using the ramps to gain access to culverts, but there remains no evidence on passage efficiency or the effective operational range of this solution. As such, flexible rubber ramps should only be deployed with extreme caution as short-term mitigation for climbing species, and strict requirements for performance monitoring until data are available to determine its effectiveness. Where it is shown to be ineffective at providing passage for the target species it should be removed and replaced with a more effective ramp type or the structure replaced.

The alternative ramp designs described above have a strong evidence base and deployment of those solutions ahead of rubber ramps would be consistent with the application of best available information as required by the NPS-FM.

Figure I-1: Example of a rubber ramp deployed at a perched culvert in combination with mussel spat ropes.

Synthesis & recommendations

There is a strong weight of evidence demonstrating that **smooth substrate ramps should be avoided** (Baker and Boubée 2006; Jellyman et al. 2016). Presently, a Miradrain™ substrate has consistently demonstrated the best performance for enhancing upstream passage of small-bodied species (Baker and Boubée 2006; Baker 2014; Jellyman et al. 2016; Franklin et al. 2021).

Based on the current evidence base, and presuming a Miradrain™ substrate, we recommend that **maximum ramp slope should be 15° and ramp length at this slope should not exceed 1.5 m**. This is based on the significant reduction in passage success observed at higher slopes and/or ramp lengths across all studies. Evidence suggests that slopes of closer to 5˚ may be required to achieve high passage rates over ramps of up to 3 m, but slopes of c. 2–3˚ may provide passage up to distances of c. 7 m (Hicks et al. 2008). Ramps >1.5 m will require resting pools at each 1.5 m interval. See Section 5.5.2 for guidelines on designing resting pools.

A ramp of 1.5 m at a slope of 15° corresponds to a fall height (h) of 0.39 m, while a 3 m ramp at a 5˚ slope corresponds to h = 0.26 m and for a 7 m ramp at 3˚ h = 0.37 m.

As such, **artificial substrate ramps are most suited to short-term mitigation in situations where the maximum fall height is up to 0.4 m**. For fall heights >0.4 m or long-term mitigation, rock ramp designs should be used, or a trade-off will have to be made against fish passage efficiency.

Fixed ramps are considered to have a higher passage efficiency across a wider range of species and life stages compared to floating ramp designs (Baker and Boubée 2006; Fake 2018). However, floating ramps can provide some effective short-term mitigation (Fake 2018). An important design consideration for floating ramps is that the slope will vary with the downstream water level and so passage rates are likely to be variable over time as the ramp moves. Similarly, the effective ramp length of fixed ramps will vary over time with differing tailwater levels. Ramp gradient and ramp length will be greatest for floating and fixed ramps, respectively, under low tailwater levels. Consequently, ramp installation should be designed to ensure the ramp is within the optimum operating range (e.g., ramp slope ≤15°) under conditions when the tailwater level is low and the head drop is maximised.

One of the critical limitations of all these ramps is that they do not provide passage for adult life stages of most fish species. As such, they are only recommended as a short-term/temporary mitigation solution until alternative solutions that cater for all species and life stages (e.g., rock ramp fishways) can be installed.

Appendix J Remediation evidence synthesis: Part 2 - baffles

Overview

A range of baffle types and configurations have been proposed and tested for enhancing fish passage through culverts (Frankiewicz et al. 2021). These include various weir style baffles, spoiler baffles, offset baffles, vertical baffles, corner baffles, and longitudinal beams. Much of the work to date has focused on characterising the hydraulics of different baffle types (Ead et al. 2002; Feurich et al. 2011; Duguay and Lacey 2015; Zhang and Chanson 2018; Magaju et al. 2021). However, there are an increasing number of laboratory and field studies evaluating fish behaviour and passage efficiency associated with different baffle types and configurations (Macdonald and Davies 2007; Franklin and Bartels 2012; Amtstaetter et al. 2017; Enders et al. 2017; Goerig et al. 2017). Evidence from these studies demonstrates that the effectiveness of different baffles for enhancing fish passage cannot be determined based on hydraulic assessment alone as the solutions that typically generate the lowest average water velocities often also create hydrodynamic conditions (e.g., large recirculating zones) not conducive to unimpeded upstream movement of fish (Feurich et al. 2011; Khodier and Tullis 2014; Duguay et al. 2018; Cabonce et al. 2019; Magaju et al. 2023).

To underpin the recommendations made in these guidelines, we undertook a rapid systematic evidence synthesis to review and summarise the evidence on the effectiveness of different culvert baffle designs for improving the abundance and diversity of small-bodied fish successfully migrating upstream through culverts. Full details of the evidence synthesis are reported in Franklin and Baker (In review) and are summarised here.

Methodology

We followed the ROSES protocol (Haddaway et al. 2018) for evidence synthesis, which is designed specifically for systematic reviews and systematic maps in the field of conservation and environmental management.

A systematic search of relevant literature was conducted to identify studies on the effectiveness of different types of culvert baffle for improving fish passage. Two electronic databases were searched (Web of Science Core Collection and ProQuest Earth, Atmospheric & Aquatic Science Collection) using a comprehensive search strategy. Search strings included the following keywords and synonyms: "culvert", “pipe”, “baffle”, “remediation”, “fish”, "fish passage", "fish migration", and "connectivity". Because we wanted to take an inclusive approach to evidence collation for this synthesis, an additional search for grey literature using the same search strings was also conducted in Google Scholar. Further supplementary searches included bibliographic searches and citation tracking to check for further studies that may have been missed in the electronic sources. Fish passage practitioners in New Zealand were also contacted via the national Fish Passage Advisory Group mailing list to ask for any local studies or unpublished data that may be relevant to the review.

The search results were initially screened based on title and abstract content to identify potentially relevant studies. Selected studies were then subjected to full-text screening. Eligibility criteria used to determine whether a study was relevant for inclusion in the rapid evidence synthesis included:

A. Study design: only primary research studies that assessed the effectiveness of different culvert baffle designs for improving fish migration through culverts were included. Experimental and observational, and laboratory and field-based studies were included.

B. Population: studies were required to be focused on improving upstream migration or movements of freshwater fishes through culverts. An emphasis was placed on identifying studies relevant to small-bodied (<150 mm) diadromous species and/or life stages, but studies on the upstream migration of other fish species/life stages were also included.

C. Intervention: only studies that evaluated the effectiveness of different types of culvert modifications intended to facilitate upstream movements of fish through the culvert barrel were included. Studies that included multiple interventions were included if they allowed for comparison of the effectiveness of the individual modifications. Studies that addressed other types of fish passage interventions at culverts (e.g., to overcome drops at culvert outlets) or at other structure types were excluded from this review.

D. Comparator: studies that compared the effectiveness of different types of culvert modifications with each other or with unmodified culverts were included. We excluded studies that only quantified hydrodynamics.

E. Outcome measures: a range of different performance measures were included to reflect the diverse ways in which fish passage success can be evaluated. Studies that quantified fish passage efficiency, passage success/failure, passage duration, and changes in upstream species richness or abundance were included, as were studies that addressed fish behaviour and hydraulic conditions (in combination) within the culvert barrel relevant to fish passage.

F. Language: only studies published in English were considered. We acknowledge that this represents a bias in our study as it will not capture non-English sources, but it was outside the scope of this project to translate and use non-English language texts.

G. Publication date: no restriction on publication date was included.

The following criteria were used to assess the risk of bias in each individual study included in the review:

A. Performance bias: it was determined whether the interventions were implemented consistently and whether the control/before and impact/after groups received similar levels of attention.

B. Detection bias: each study was evaluated to establish the risk that there were differences in the methods used to measure outcomes between the groups.

C. Confounding bias: the existence of factors that are related to both the exposure and the outcome that were not adequately controlled for in the analysis was identified.

D. Reporting bias: the risk of reporting bias was assessed by examining whether all the pre-specified outcomes were reported whether the analyses were conducted as planned and set out in the methodology, and whether there was evidence of selective reporting of outcomes, i.e., not all outcomes were reported fully.

E. Other biases: all studies were assessed to evaluate whether any other sources of bias may exist that were specific to the study design or context.

Greatest weight was placed on evidence from the peer-reviewed literature, acknowledging the rigorous review process these sources are subject to.

To reduce observer bias, a random sub-set of ten search results were independently reviewed to check for consistency of selection.

Results

A total of 24 peer-reviewed journal articles, four guideline documents, four reports, three theses, and one unpublished dataset met the criteria for inclusion in the evidence synthesis. Half of the studies included in the review were laboratory-based, while 34% were field-based studies. The remaining studies included in the review were guideline documents or reviews that included evidence summaries from laboratory and field-based studies. Just over three quarters of the included studies were based on experimental studies (laboratory and field-based) and 17% of studies included observational field studies.

Weir style baffles

Weir baffles take a variety of forms, for example standard weir baffles, slotted or notched weir baffles, and flexi baffles, but fundamentally involve the installation of horizontally oriented baffles across the base of the culvert creating a series of pools and weirs that fish must navigate (Figure J-1). Weir style baffles can be installed in box or pipe culverts, but most studies of weir style baffles included in this synthesis were focused on pipe culverts. Most biological studies on the performance of weir baffles have focused on salmonid species (e.g. Olsen and Tullis 2013; Khodier 2014; Enders et al. 2017; Duguay et al. 2018; Duguay et al. 2019). Attraction (as measured by passage attempts) appears to be lower with weir baffles compared to without in these studies (Enders et al. 2017; Duguay et al. 2019), but overall passage success typically increased compared to a bare culvert/flume (Olsen 2011; Khodier 2014; Enders et al. 2017; Duguay et al. 2019). However, several studies also identified delays in passage associated with weir baffles compared to both control conditions and other baffle types (Enders et al. 2017; Duguay et al. 2019) and reorientation of fish away from the upstream direction in areas of flow reversal associated with the baffles (Khodier and Tullis 2014; Duguay et al. 2018).

There have been few studies of passage efficiency provided by weir baffles for New Zealand or small-bodied fishes more generally. Feurich et al. (2012) reported substantial delays in upstream passage for īnanga associated with several different weir style baffles and concluded that they were not suitable for benthic species such as the bully species (*Gobiomorphus* spp.) found in New Zealand due to the absence of a continuous pathway along the culvert base. As was observed in the salmonid studies, the delays in movement were caused by disorientation of fish in the recirculation zones that develop between each baffle (Feurich et al. 2012). There have also been several case studies of flexi baffles in New Zealand, but in most cases the results are inconclusive due to the confounding effect of multiple remediation measures being applied concurrently and inconsistencies or flaws in sampling design and methodologies (e.g. Olley et al. 2024). At one culvert (136 m long) fitted only with flexi baffles, a mark-recapture study using īnanga whitebait showed 0% passage success at the end of the 48-hour trial period (NIWA unpublished data). The maximum distance travelled within the culvert during the trial was 45 m, but most fish (66%) failed to progress further than 12 m up the culvert. While this was a relatively complicated site and so achieving high overall passage efficiency was unlikely, the rate of progress through the lowest gradient section (2˚) at the start of the culvert is exceptionally poor relative to equivalent mark-recapture studies with the same species for other baffle designs (Franklin and Bartels 2012; Franklin et al. 2018; Baker et al. 2024a; Franklin et al.

2024). In an additional trial, extended trapping upstream of the same culvert has shown low numbers of īnanga passing the culvert after baffle installation (Olley 2020), but no comparable before data were collected and no information on fish size was provided to determine whether there was a size bias in successful fish. However, īnanga passing the culvert were pigmented, feeding fish indicating they were post-whitebait. It is possible that in a culvert this length, that low light may have been a confounding impediment to movement (Jones et al. 2017; Keep et al. 2021).

A comparative study of īnanga passage through an experimental 6 m culvert with and without flexi baffles indicates higher passage rates with baffles present across a range of gradients (Olley et al. 2024). However, these results appear to be based on a single replicate under each treatment and no information is reported on fish sizes, water temperatures, nor timing of trials. Preliminary results from field-based evaluations of flexi baffles indicate that there is a bias in passage success with fish size, with larger fish having a higher success rate (Easton 2023). The same bias in the size of fish able to pass flexi baffles was also reported by Olley and Olley (2022) for a steep gradient (4–9%) culvert.

Figure J-1: Examples of weir style baffles including slotted baffles (Left), flexi-baffles (Centre) and standard weir baffles (Right).

A slight variation on the standard application of weir baffles within circular culverts is to install the baffles with the crest rotated 10–20˚ from horizontal with the objective of improving the passage of sediment (Olsen 2011). These are sometimes referred to as either sloped baffles or corner baffles. Olsen (2011) observed statistically significant improvements in passage efficiency for brown trout (200–350 mm FL) compared to a bare culvert. Furthermore, passage was achieved across a higher range of slopes and was more consistent with variations in flow. Newbold et al. (2014) also evaluated the effects of sloped baffles on the upstream passage of European eels (mean length 439 mm TL) and found post-entry passage efficiency was no different between the baffled and bare culvert, but because entrance efficiency was higher for the culvert with baffles overall passage success was greater with the baffles in place.

Spoiler baffles

Spoiler baffles take the form of blocks set in a staggered configuration that slow and disperse the flow of water near to the culvert bed (Figure J-2). They can be deployed on the base of pipe or box culverts, which were represented equally in the studies included in this synthesis. Various spoiler baffle arrangements are available, but the geometry and spatial configuration of the baffles must be designed to take account of the needs of the target fish species or communities.

Figure J-2: Examples of spoiler baffles.

Spoiler baffles have been the recommended baffle design in New Zealand for the last 15 years (Stevenson et al. 2008; Franklin et al. 2018). This recommendation arose following comparative assessment of a range of different baffle types for their suitability for enhancing the passage of native fishes in New Zealand (Stevenson et al. 2008; Feurich et al. 2011; Feurich et al. 2012). Early hydraulic studies demonstrated similar performance across multiple baffle types in terms of their impact on bulk water velocity and water depth (Ead et al. 2002). However, more recent biological studies have repeatedly demonstrated that the suitability and effectiveness of different baffle types cannot be determined solely from their impact on bulk hydraulic parameters (Feurich et al. 2012; Khodier and Tullis 2014; Enders et al. 2017; Duguay et al. 2018; Vowles et al. 2019; Magaju et al. 2023).

Spoiler baffles were identified as a preferred design for New Zealand species for the following reasons (Stevenson et al. 2008; Feurich et al. 2012):

- Lower velocity zones are created allowing for improved upstream passage of fish.
- They allow for a continuous pathway along the bed of the culvert for the passage of benthic species.
- They generate low velocity resting zones within the baffle array.

- There is sufficient space within the baffle field for large-bodied species and life stages to pass through.

Comparative studies have reported much lower passage delay for fish through spoiler baffle arrays when compared to weir baffles (Feurich et al. 2012; Duguay et al. 2019). It was hypothesised that this reflected the different hydrodynamics created by the two different baffle types, with the large-scale recirculation zones created by weir baffles resulting in disorientation and delays in upstream movements of fish. This hypothesis is reinforced by observations of similar impacts on successful fish passage arising from reversing flows by other authors (Khodier and Tullis 2014; Duguay et al. 2018; Cabonce et al. 2019; Magaju et al. 2023).

Macdonald and Davies (2007), Franklin and Bartels (2012) and Patchett (2023) all reported significant improvements in passage success of small-bodied galaxiid fishes compared to control conditions in field applications of spoiler baffles under similar flows. In contrast, Vowles et al. (2019) observed reduced passage success compared to a non-baffled control for river lamprey (*Lampetra fluviatilis*), a Northern Hemisphere species. It is important to note that river lamprey have significantly different morphology, behaviour and movement capabilities (cannot climb) compared to the native pouched lamprey (*Geotria australis*) and the results of Vowles et al. (2019) are not transferable to New Zealand lamprey.

Recent experimental flume-based studies of fine scale native fish movements within different spoiler baffle configurations sought to understand how fish interacted with the different hydrodynamic characteristics and to refine design recommendations (Magaju et al. 2021; Magaju et al. 2023). It was observed that īnanga and kōaro sought out lower water velocities (<0.5 m s^{-1}) during upstream movement and that they preferentially used small-scale (i.e., less than the body size of the fish) vertically oriented wake eddies for resting (Magaju et al. 2023). Based on these findings of how fish use the different hydrodynamic features within a spoiler baffle field, Magaju et al. (2023) concluded that halving the length of the standard spoiler baffle used to date in New Zealand may improve passage efficiency by increasing the area of wake eddy zones available that allow fish to rest while remaining oriented into the main direction of flow. However, this new baffle configuration has not yet been tested at full scale or in the field.

Vertical baffles

Vertically oriented baffles are affixed to the side walls of culverts rather than the culvert base (Figure J-3). In box culverts they typically span most of the culvert height, while in pipe culverts, they frequently only extend halfway up the pipe. In pipe culverts, they are typically installed only on one side of the pipe, whereas in box culverts they may be installed on both side walls. Various designs and configurations of vertical baffle have been proposed, but there is relatively little literature on their performance.

Figure J-3: Examples of vertical baffles. Photo credit: Tim Marsden, Australasian Fish Passage Services.

Enders et al. (2017) compared the effectiveness of vertical and horizontal (weir style) baffles in a square flume for Alewife (*Alosa pseudoharengus*) and Brook trout (*Salvelinus fontinalis*) and found more passage attempts were made with vertical baffles compared to horizontal baffles, but that Alewife achieved a higher maximum distance at first attempt with weir baffles under low flows. At higher flows, the effectiveness of weir baffles for Alewife was substantially reduced, while the effectiveness of vertical baffles was largely unchanged resulting in no difference in passage success between the baffle types. There was no difference in passage success for Brook trout between the two baffle types at first attempt, but passage success was higher with vertical baffles for repeat passage attempts (Enders et al. 2017).

Marsden (2015) and Amtstaetter et al. (2017) tested vertical baffles with small-bodied Australian fishes in a box and circular culvert respectively. Both studies demonstrated a 30–40% improvement in passage efficiency compared to control treatments without baffles; in the case of Amtstaetter et al. (2017) this was for small-bodied galaxiids including īnanga. While the species tested by Marsden (2015) were not similar to New Zealand species, the most common fish were 20–40 mm Empire gudgeon (*Hypseleotris compressa*) demonstrating their potential effectiveness for small-bodied species. Of note were the observations from Marsden (2015) that wider baffles (300 mm) generated large flow reversals behind the baffles, and this led to reorientation of fish to the downstream direction and caused downstream sweep of fish as they tried to pass the next baffle.

Offset baffles

The standard offset baffle configuration (Figure J-4) consists of a series of low baffles installed on the base of a culvert including short baffles affixed perpendicular to one culvert wall and longer baffles fixed at 30˚ to the opposite wall (Rajaratnam et al. 1988; Kapitze 2010). While once widely deployed in North America and Europe, it has largely been replaced by other designs in recent decades (Kapitze 2010). Baffle spacing and size have a significant influence on hydraulics and water velocities through the slot between baffles (Rajaratnam et al. 1988).

Kapitze (2010) reported increases in the diversity of fish species captured upstream of a culvert fitted with offset baffles in Australia, including some small-bodied species, but more generally there appears to be little in the way of evidence describing their performance for small-bodied species.

Figure J-4: Examples of offset baffles. Source: Kapitze (2010).

Triangular corner baffles

Small triangular corner baffles were proposed as a means of facilitating upstream passage of small-bodied fishes in box culverts in Australia (Cabonce et al. 2017; Cabonce 2018; Cabonce et al. 2019). The baffles are isosceles triangles that are affixed to the bottom corner of box culverts (Figure J-5). Most of the work to date has focused on characterising the hydraulic performance of these corner baffles (Cabonce et al. 2017), with only two studies evaluating the performance with fishes (Watson et al. 2018b; Cabonce et al. 2019). The hydraulic studies indicate that they generate low velocity zones immediately upstream and downstream of the baffles that have the potential to be utilised by small-bodied fish (Cabonce et al. 2017; Cabonce 2018). However, they also generate flow reversals in the lee of the baffles that can disorientate some fish and prevent them from progressing upstream (Cabonce et al. 2019). Inclusion of a hole in the baffle mediated this hydraulic effect to some extent (Cabonce 2018). However, while biological testing of triangular corner baffles with juvenile silver perch (*Bidyanus bidyanus*) in Australia showed some improvements in swimming endurance relative to the control treatment (Cabonce et al. 2019), in the case of six other small-bodied Australian species/life stages, triangular corner baffles (both ventilated and non-ventilated) provided no performance benefit compared to a control (and in fact reduced performance for some species) and achieved 0% passage success over a 12 m flume (Watson et al. 2018b).

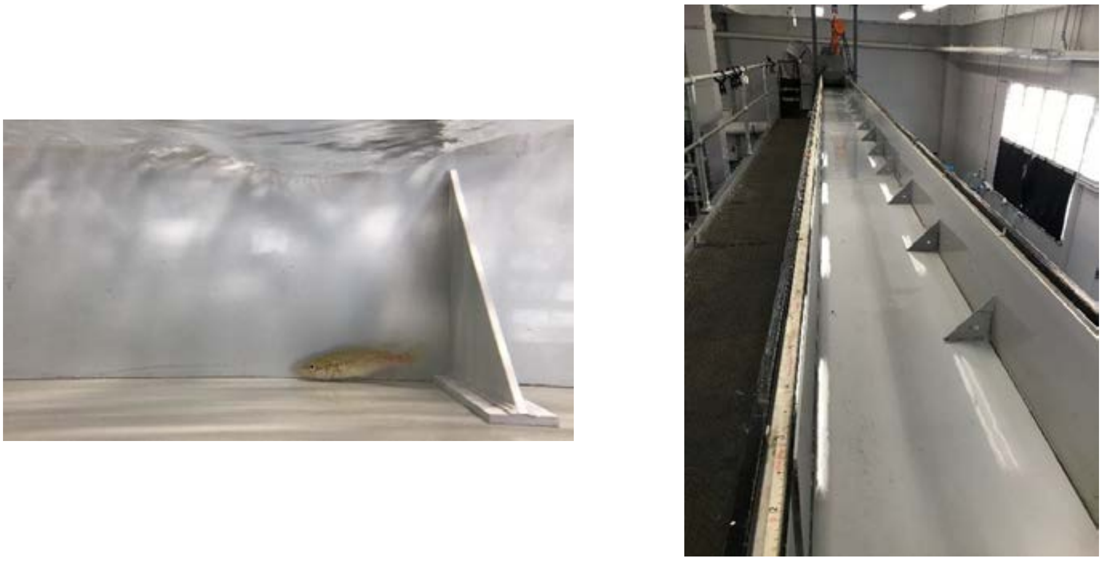


Figure J-5: Example of corner baffles. Photo credit: Craig Franklin & Rebecca Cramp, University of Queensland.

Longitudinal baffles

Longitudinal baffles take the form of beams that run along the length of the structure and have been tested by Watson et al. (2018b) in a rectangular flume (i.e., box culvert) for a selection of small-bodied and juvenile Australian fishes. The concept is based on extending the low velocity boundary zones along culvert walls that fish are observed to exploit naturally. Watson et al. (2018b) experimentally compared the performance of three beam designs: square beams (50 × 50 mm), semi-circular beams (55 mm radius), and a ledge (50 mm width). The beams were installed longitudinally along the side walls of a rectangular flume 50 mm above the flume bed. Swimming endurance was increased relative to the control (no beam), but the most beneficial design of longitudinal baffle varied between individual species. Overall passage success was significantly higher across all beam designs relative to the control. While there was no significant difference in the overall performance between the different beam designs, again the benefits of different beam designs for improving passage success varied between species.

Figure J-6: Examples of longitudinal baffles. Photo credit: Craig Franklin & Rebecca Cramp, University of Queensland.

Alternating baffles

Alternating weir baffles (that do not extend across the full culvert width) have been installed at several sites in New Zealand but have been subject to limited testing. One application was shown by Patchett (2023) to improve passage rates of migratory galaxiids following installation in a box culvert, but significantly decreased passage success of benthic *Gobiomorphus* sp.

Figure J-7: Example of alternating baffles. Photo credit: Cindy Baker.

Synthesis

The overall weight of evidence suggests that for the range of species and baffle designs tested, the addition of baffles (of most designs) generally increases the overall passage success rate compared with a control treatment with no baffles in both pipe and box culverts. There remains a lack of comparative studies that robustly compare the performance of different baffle categories/types across both box and pipe culverts. However, biological studies have repeatedly demonstrated that the suitability and effectiveness of different baffle types for passing fish cannot be determined solely from their impact on bulk hydraulic parameters and also that overall effectiveness cannot be determined based on simple passage efficiency metrics (Feurich et al. 2012; Khodier and Tullis 2014; Enders et al. 2017; Duguay et al. 2018; Vowles et al. 2019; Magaju 2023).

There is, however, an emerging body of evidence that focuses on different baffle designs suitable for facilitating the upstream passage of small-bodied species/life stages through both pipe and box culverts (e.g. Feurich et al. 2012; Franklin and Bartels 2012; Marsden 2015; Amtstaetter et al. 2017; Watson et al. 2018b). While early research on baffle hydraulics (e.g. Rajaratnam and Katopodis 1990) indicated that weir style baffles were most effective at reducing bulk water velocities, a consensus is emerging in the scientific literature and fish passage guidelines that alternative baffle styles are better for improving the upstream passage of small-bodied fishes (Stevenson et al. 2008; O'Connor et al. 2017a; Franklin et al. 2018; Watson et al. 2018b; Magaju 2023). Turbulent structures in flowing waters are known to influence fish behaviour (Liao 2007; Lacey et al. 2012). Baffle arrays generate a range of different hydraulic conditions and turbulent structures depending on their design (Rajaratnam et al. 1990; Zhang and Chanson 2018). Multiple lines of evidence are emerging that indicate that the nature of these turbulent structures has a significant impact on the passage of fish through baffle arrays. Eddies and areas of flow reversal that are larger in scale than the size of the fish have been observed to cause individuals to reorientate away from the upstream direction (Feurich et al. 2012; Marsden 2015; Enders et al. 2017; Duguay et al. 2018; Watson et al. 2018b; Magaju et al. 2023). This causes delays in upstream movement as fish become disoriented in recirculation zones (Enders et al. 2017; Duguay et al. 2018) and downstream sweep of individuals as they attempt to re-enter the bulk flow (Feurich et al. 2012; Marsden 2015; Cabonce et al. 2019; Magaju 2023). In contrast, individuals have been observed to exploit small-scale (i.e., less than the size of the fish) turbulent eddies and wake zones to hold station and conserve energy (Liao et al. 2003a; Liao 2004; Magaju et al. 2023) and actively utilise boundary layers to facilitate upstream movement (Cabonce et al. 2017; Watson et al. 2018b).

The weight of evidence suggests that passage efficiency for larger pelagic fishes is generally higher using weir baffles compared to having no baffles in place (e.g. Olsen 2011; Khodier 2014; Enders et al. 2017; Duguay et al. 2019). However, there is also strong evidence to suggest that the large recirculation zones created by weir baffles impose significant delays on upstream movements across multiple species and that this effect is likely greatest for small-bodied fish (Feurich et al. 2012; Khodier and Tullis 2014; Duguay et al. 2018; Magaju 2023). Due to the preference of benthic species for a continuous movement pathway along the bed, horizontal weir baffles, including flexi baffles, are unlikely to be suitable for providing effective passage. **Presently, the evidence base in support of using weir style baffles (including flexi baffles) for improving the upstream passage of New Zealand’s native fishes remains weak relative to other designs and they are not recommended for use.** Weir style baffles are also not recommended for use in Australia where they are catering for similar small-bodied species (Kapitze 2010; O'Connor et al. 2017a).

The weight of evidence suggests that designs that minimise the generation of large recirculation zones while maximising the area of low velocity boundary layers are preferable for achieving unimpeded passage of small-bodied fishes (e.g. Watson et al. 2018b; Magaju et al. 2023). **As such, the weight of evidence indicates that spoiler baffles, vertical baffles and longitudinal baffles are best suited to enhancing the passage of small-bodied species in New Zealand.** Presently, the strongest evidence base is for the use of spoiler baffles, although results from Australia indicate that vertical baffles may have application in a New Zealand context, but are yet to be tested here. Kapitze (2010) and O'Connor et al. (2017a) recommend application of vertical baffles in box culverts for facilitating the upstream passage of small-bodied fishes in Australia. Based on the results of Watson et al. (2018c), it is likely that longitudinal baffles have the potential to be beneficial for small-bodied New Zealand fishes. However, they have not yet been tested for New Zealand fishes. A potential benefit of the longitudinal baffle design is that they appear to generate enlarged reduced velocity zones along the culvert wall that can be utilised by fish, but without generating larger scale turbulence or recirculating zones that have been observed across multiple studies to disorientate fish and delay upstream movements (Li et al., 2021, Sanchez et al., 2020). To date, the passage efficiency of longitudinal baffle designs has not been tested in circular culverts, but physical modelling of a longitudinal baffle in a circular culvert indicates the hydraulic performance is transferable (Li et al., 2021).

There is currently weak evidence either in support or against the use of alternating baffles for New Zealand fishes. Positive aspects of the design include the availability of a continuous pathway along the bed of the culvert for benthic species and the availability of low velocity resting zones. Possible negative aspects include the presence of large-scale eddies and flow reversals that have been observed to disorientate small-bodied fishes and delay upstream movements. The current evidence on corner baffles indicates they are likely unsuitable for use in New Zealand. Offset baffles may provide some benefit, but the current weight of evidence suggests that they are likely sub-optimal for use in New Zealand.

Summary

The overarching goal of fish passage remediation is generally to maximise the proportion of each species arriving at a structure that can pass and to minimise the time it takes for them to pass. There is sufficient weight of evidence to indicate that the addition of baffles mostly generates a positive effect on overall passage success rates compared with doing nothing in both pipe and box culverts. However, performance is highly variable between species and among different baffle types.

The influence of turbulence has emerged as an important control on the speed of passage, with large recirculation areas between baffles causing fish to become disoriented and delaying upstream progress, particularly for small-bodied fishes. The current weight of evidence suggests, therefore, that weir style baffles, or any baffle configuration that generates large recirculating zones, should be avoided when the target is to optimise upstream movements of small-bodied species. However, there is emerging evidence that small-scale wake eddies in spoiler baffle and vertical baffle arrays can be exploited by small-bodied fish to increase the efficiency of their upstream movements.

Recommendations

For small-bodied fishes, the current weight of evidence indicates that weir style baffles can cause significant delays in upstream movements due to the large recirculation zones they generate between baffles. Consequently, **weir style baffles of all types and corner baffles should be avoided where small-bodied fish are required to pass**.

The current evidence suggests that baffles that minimise the generation of large eddies and recirculation zones (e.g., spoiler baffles, vertical baffles, longitudinal baffles) and maintain a continuous pathway along the culvert base are more effective for facilitating upstream passage while also minimising delays in movements. At present, there is sufficient evidence available to indicate that **spoiler baffles and vertical baffles are the recommended option for restoring passage of small-bodied fishes through culverts in New Zealand.**


NIWA
Taihoro Nukurangi
niwa.co.nz/fishpassage