Environmental assessments in support of shellfish farming in Albany, Western Australia

South Coast Aquaculture Development Zone

R-1757_00-2

July 2021

BMT Commercial Australia Pty Ltd
Level 4, 20 Parkland Road
Osborne WA 6017
Australia
PO Box 2305, Churchlands, WA 6918

Tel: +61 8 6163 4900

ABN 54 010 830 421

www.bmt.org

ADDENDUM

This report, *Environmental Assessments in Support of the South Coast Aquaculture Development Zone (SCADZ)*, Albany, Western Australia, July 2021 supersedes the report of the same name, published in March 2021.

This July 2021 version includes substantive amendments following extensive consultation with industry and a site visit on 10 June 2021. The new information was used to update the science behind the original carrying capacity model, which in turn provided an opportunity to re-run the model using less-conservative, but more precise assumptions.

Changes to the assumptions included (a) reduced shellfish clearance rates and exposure times, (b) decreased shellfish feeding efficiency and (c) new base line information based on historical shellfish aquaculture operations and the associated level of biofouling.

On aggregate, this resulted in increased baseline food (phytoplankton) availability and a material reduction in shellfish feeding and clearance rates. When remodelled, this served to increase the theoretical carrying capacity of the system of Oyster Harbour and King George Sound, with a minor reduction in the targeted production for Princess Royal Harbour (this report).

The following report has consequently been updated to reflect this via changes to the Executive Summary, and Sections 5, 6 and 8. The content of the changes is detailed below:

Amendments / New material

- Updates to the baseline model to include actual and extrapolated biomasses of farmed shellfish and bio-fouling, rather than assumed biomasses.
- A change in phytoplankton concentration from 0.12 to 0.50 μg.chlorophyll-a/L in response to the updated oceanic boundary condition.
- New biomasses in PRH and KGS following a change in zoning arrangements.
- Inclusion of bio-fouling in the model, equivalent to 50% of the farmed biomass under baseline and future production scenarios.
- Reduction in the assumed clearance efficiency of shellfish from 100% to 80% for all species.
- Reduction in the percentage of time *Saccostrea glomerata* are submerged from 100% to 75%, to match the estimated exposure time over their three-year grow-out cycle.
- Reduction in the clearance rates of *Pinctada imbricata fucata* modelled from 11, 25 and 103 L g^{-1} h^{-1} to 5, 15 and 30 L g^{-1} h^{-1} based off updated in-situ data for this species.

As a final comment, BMT wishes to emphasise that the results presented herein are based on a one-dimensional hydrodynamic model, coupled to a simple ecosystem model. It relies on several assumptions, many of which are untested, or if not characterised by significant uncertainty (despite the opportunity to improve them in this iteration of the report).

The approach to dealing with the uncertainty was to err on the side of caution, and use numbers at the upper or lower end of the range for each respective variable – whichever presented the greatest risk to the environment. The interim carrying capacity targets presented in the body of the report are therefore highly conservative, and likely lower than the SCADZ can sustain, before realising any material impacts to the local ecology.

Ostensibly, the interim targets serve as a conservative starting point for the safe expansion of the industry, pending collection of further data and ultimately, the reassessment of the numbers. It is recommended that any expansion of the industry beyond the numbers herein, is undertaken with caution, and following the analysis of environmental monitoring and shellfish production data, together with the validation of the ecosystem model and its assumptions.

BMT
13 July 2021

Document Control Sheet

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Quality Assurance

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BMT Commercial Australia Pty Ltd has prepared this report in accordance with our Integrated Management System, certified to OHSAS18001, ISO14001 and ISO9001

Status

This report is 'Draft' until approved for final release by the Project Director (or their authorised delegate) as indicated below by signature. A Draft report may be issued for review with intent to generate a 'Final' version but must not be used for any other purpose.

Approved for final release:

Director (or delegate)
Date: 13/07/2021

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Acronyms

Acronym	Full list
ACWA	Aquaculture Council of Western Australia
ANZECC	Australian and New Zealand Environment and Conservation Council
APA	Albany Port Authority
ASP	Amnesic shellfish poisoning
ASQAP	Australian Shellfish Quality Assurance Programme
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
BCH	Benthic communities and habitats
CFP	Ciguatera fish poisoning
Chl-a	Chlorophyll-a
CoA	City of Albany
DBCA	Department of Biodiversity, Conservation and Attractions
DO	Dissolved Oxygen
DOC	Dissolved organic carbon
DoH	Department of Health
DPaW	Department of Parks and Wildlife
DPIRD	Department of Primary Industries and Regional Development
DPLH	Department of Planning, Lands and Heritage
DSP	Diarrhetic shellfish poisoning
DWER	Department of Water and Environmental Regulation
EPA	Environmental Protection Authority
FB	Frenchman Bay
FLUPSY	Floating upweller system
GIS	Geographical information systems
HAB	Harmful algal bloom
IMS	Introduced marine species
KGS	King George Sound
LAU	Local assessment unit
MI	Mistaken Island
NAGD	National Assessment Guidelines for Dredging
NSP	Neurotoxic shellfish poisoning
OGA	Ocean Grown Abalone
OH	Oyster Harbour
ppt	Parts per thousand
POC	Particulate organic carbon
PRH	Princess Royal Harbour
PSP	Paralytic shellfish poisoning
SCADZ	South Coast Aquaculture Development Zone
South Coast NRM	South Coast Natural Resource Management
TOC	Total Organic Carbon
TON	Total organic nitrogen
TOP	Total organic phosphorous
WA	Western Australia

Acronym	Full list
WASQAP	Western Australia Shellfish Quality Assurance Program
ZoHI	Zone of High Impact
ZoI	Zone of Influence
ZoMI	Zone of Moderate Impact

Executive Summary

Background

The Department of Primary Industries and Regional Development (DPIRD) proposes to develop a new aquaculture zone in nearshore waters adjacent to the regional centres of Albany and Esperance, hereafter referred to as the South Coast Aquaculture Development Zone (SCADZ). The SCADZ will be developed in stages, commencing initially in Albany (Figure 1.1) before potentially expanding to sites in Esperance. The proposed Albany development comprises of several leases in Oyster Harbour (OH), one in Frenchman Bay, two at Mistaken Island and a new lease in south-eastern Princess Royal Harbour (PRH). The declaration of the SCADZ in Albany will occur in two stages. OH was declared in August 2020, with PRH and King George Sound (KGS) to follow once the stakeholder consultation phase is complete.

To support the development of the SCADZ, DPIRD engaged BMT to conduct an assessment of the potential impacts of the proposed development on the local marine environment. At the request of DPIRD, this report focuses primarily on the Albany environment. The exception is the benthic communities and habitat (BCH) assessment, which was conducted for the Esperance and Albany environments.

Benthic communities and habitats assessment - Albany

The potential for impacts to BCH was assessed using the criteria set out in EPA (2016a, b). Cumulative losses were determined based on separate loss assessment units (LAUs): Princess Royal Harbour, Inner King George Sound and Oyster Harbour.

Existing information on BCH was examined within each LAU to determine the extent of coverage prior to European settlement, subsequent losses post European settlement and the losses expected due to the SCADZ development.

The potential for shellfish farming to impact BCHs was considered low, and where unavoidable, restricted to the effects of 'piercing' or 'smothering' due to the placement of posts, anchors and artificial substrates. This may be mitigated by the careful placement of infrastructure to avoid areas of significant BCH cover. The potential for indirect effects due to shading or benthic nutrient enrichment was also considered very low, and where present, fully recoverable assuming application of best-practice aquaculture protocols.

The potential for permanent (>5 years recovery) and/or recoverable losses (<5 years recovery) were calculated based on the most likely positioning and configuration of aquaculture infrastructure (i.e. posts, longlines and baskets). Permanent losses of BCH were shown to be negligible at <0.1% and recoverable losses minimal at <5% of the LAUs. The ecological function of BCH in Albany was therefore considered at low risk, particularly if best practice operations and management strategies are followed.

Benthic communities and habitats assessment - Esperance

The LAU for Esperance was newly derived due to a lack of precedent. Habitat information was compiled using the same approach as in Albany. The Esperance assessment assumed exclusive allocation of leases to the abalone industry. For operational reasons, the artificial habitats used for ranching (ABITATs) are placed exclusively on sandy substrates, at least 50 m from seagrasses or macroalgal communities. As such, future losses attributable to the SCADZ were presumed to be zero. At the time of the assessment it was uncertain whether shellfish farming would be extended to Esperance; however, assuming similar operations are adopted (as in Albany), permanent losses to BCH were assumed to be <0.1%. The proposed expansion of the SCADZ to Esperance therefore poses little risk to the ecological function of BCH in the local region.

Carrying capacity

For this study, BMT employed AquaDEEP, a one-dimensional hydrodynamic and ecosystem model to estimate the carrying capacity of OH, KGS and PRH. Carrying capacity was defined as the standing biomass that could be sustained, without reducing regional phytoplankton biomasses to levels below 1 µg.chlorophyll-a/L, which is the threshold for healthy oligotrophic ecosystems in southwestern Western Australia (Brearley 2005).

Multiple operational scenarios were simulated based on present-day farming activities and future production configurations provided by DPIRD and industry. The scenarios simulated by the model were as follows:

- Oyster Harbour (*Saccostrea glomerata*)
    - Current baseline of 306 t annual production (918 t standing biomass – assuming a 3-year grow out)
    - 406 t annual production (1218 t standing biomass)
    - 456 t annual production (1368 t standing biomass)
    - 506 t annual production (1518 t standing biomass)
    - 606 t annual production (1818 t standing biomass)
    - 1206 t annual production (3618 t standing biomass)
    - 1806 t annual production (5418 t standing biomass)
- Princess Royal Harbour (*S. glomerata*)
    - Current baseline of 0 t annual production
    - 14 t annual production (40 t standing biomass)
    - 20 t annual production (60 t standing biomass)
    - 27 t annual production (80 t standing biomass)
- King George Sound (*Pinctada imbricata fucata*)
    - Current baseline of 0 t annual production
    - 68 t annual production (68 t standing biomass – assuming a 1-year grow out for *P. fucata*)
    - 104 t annual production (104 t standing biomass)
    - 139 t annual production (139 t standing biomass)
- King George Sound (*Mytilus galloprovincialis*)
    - Current baseline of 500 t annual production (500 t – assuming a 1-year grow out for *M. galloprovincialis*)
    - 736 t annual production (736 t standing biomass)
    - 854 t annual production (854 t standing biomass)
    - 972 t annual production (972 t standing biomass)

PRH will likely be used as a depuration rather than a grow-out site for OH, and as such productivity is less of a concern for this site. Production targets are still estimated for PRH however to maintain consistency with the other areas modelled and to provide a starting point in the event PRH is used for grow-out of shellfish in the future.

The model employed in this study built on previous carrying capacity assessments (Crawford et al. 1996 and Joyce et al. 2010) by incorporating calendar seasons, simple bio-energetic functions and a range of clearance rate options (minimum, medium and maximum, based on extensive literature and framing experience). Although the model was calibrated to long-term water quality trends, it lacked the data needed to validate the hydrodynamic and bio-energetics components of the model.

The modelled carrying capacity targets provided below are therefore conservative and interim, pending validation of the model and the results of ongoing monitoring. Ostensibly, however, the modelled targets provided for OH, KGS and PRH are considered safe and achievable based on the best available knowledge of regional food availability, with no expectation that they will lead to ecological impacts. We stress however, that any growth beyond these numbers should be approached with caution and subject to the results of ongoing monitoring, including chlorophyll-a and shellfish growth.

In this study, phytoplankton was depleted to <1 µg.chlorophyll-a/L based on annual productions of 506-606 t in OH, and <0.5 µg.chlorophyll-a/L based on annual productions of 20 t in PRH and 68 and 736 t in KGS, for *P. fucata* and *M. galloprovincialis* respectively. Phytoplankton concentrations are regularly <1 µg.chlorophyll-a/L in PRH and KGS, but not in OH (Thompson 2018, DWER 2020). Based on these results, modelled interim carrying capacity targets were extrapolated for each system. Targets presented below are inclusive of the standing biomasses currently farmed in the systems:

- OH: **506 t** to **606 t**
- PRH: **20 t**
- KGS: **68 t** (*P. fucata*) and **736 t** (*M. galloprovincialis*)

The interim targets serve as a conservative starting point for the safe expansion of the industry, pending collection of further data and ultimately, the reassessment of the carrying capacity targets. It is recommended that any expansion of the industry beyond the numbers presented herein, is undertaken with caution, and following the analysis of environmental monitoring and shellfish production data, together with the validation of the ecosystem model and its assumptions.

Risk of benthic nutrient enrichment

The potential for benthic nutrient enrichment was considered using the modelled bio-deposition rates coupled to the carrying capacity assessment. Bio-deposition rates were considered in the context of the local hydrodynamics and the relevant literature.

The maximum risk related to the farming of *S. glomerata* in OH, which according to the model, achieves a maximum bio-deposition rate of 8.027 grams [dry weight] per m^2 per day; a figure at least an order of magnitude lower than the bio-deposition rates demonstrated to result in minor impacts to sediments (based on the literature). Current farm practices ensure the consistent movement of aquaculture infrastructure (e.g. oyster baskets), reducing the potential cumulative impacts on benthic environments through a process of fallowing.

These results notwithstanding, the assessment of benthic nutrient enrichment was restricted to modelled rates of faecal and pseudo-faecal deposition. Management of the SCADZ should also consider the contribution of biofouling which under current farm practices is detached and disposed of to the surrounding water. Based on this, it is recommended that sediment nutrient parameters are monitored for an initial period until it can be ascertained that the impacts of biofouling removal are benign, and not at risk of exceeding the environmental quality guidelines.

Risk of harmful algal blooms

The risks posed by harmful algal blooms (HABs) in OH and KGS are known to operators, and managed under the 'conditionally approved' and 'approved' WASQAP criteria. Our risk assessment therefore focussed on PRH, which is yet to be classified. Risks in PRH were determined based on the potential for operations to effect changes to regional water quality, and particularly the characteristics likely to increase the probability of an algal bloom.

The study focused on nutrient limited algal groups (e.g. *Pseudonitzschia* sp.) and dinoflagellates. Modelling suggested the risks posed by nutrient limited groups will likely decline, whereas risks posed by dinoflagellates would remain unchanged. Risks posed by HABs were predicted to remain 'moderate' (conditionally approved) in OH, and 'low' (approved) in KGS as determined under previous classifications. The risk to PRH is pending further study, though results to date point to a 'moderate' level of risk.

It was also noted that the abundance and speciation of HABs will be affected by climate change, especially as extreme weather and fire events become more prevalent. Recent events associated with the January 2020 bush fires, for example, led to changes in the biochemistry of harmful algal groups in eastern Australia, leading to enhanced toxicity. Further work was recommended to better understand the risk posed by changing environmental conditions, and what they may mean for the industry as it evolves.

Recommendations

There remains scope to improve understanding and therefore management of the SCADZ via the following actions:

- Finetune the model and carrying capacity estimates using operational data and improved knowledge of shellfish bio-energetics (e.g. clearance rates, bio-deposition rates);
- Closely monitor the system in the initial phases of operations, cross-check the results with model outcomes (and validate and remodel as appropriate);
- Develop an interim sediment monitoring program around the aquaculture leases to evaluate the potential for enrichment due to faecal and biofouling deposition;
- Quantify the presence and abundance of HAB cysts in the sediments (primarily dinoflagellates) to better understand the underlying risk of HAB occurrences; and finally,
- Examine the potential effects of climate change to the industry, including the extent to which risks may change in the future.

1 Introduction

1.1 Background

The Department of Primary Industries and Regional Development (DPIRD) proposes to develop a new aquaculture development zone in the nearshore waters adjacent to the regional centres of Albany and Esperance, hereafter referred to as the South Coast Aquaculture Development Zone (SCADZ). The SCADZ will be developed in stages, commencing initially in Albany (Figure 1.1) before potentially expanding to sites in Esperance (Figure 1.2). This document relates primarily to development of the Albany region.

The proposed Albany component of the SCADZ will be comprised of several leases at OH, one in Frenchman Bay, two at Mistaken Island (which for the purposes of this assessment were considered as one) and a new lease in south-eastern PRH (Figure 1.1).

The consolidation of leases forms part of a DPIRD objective to support the sustainable aquaculture of edible shellfish in Western Australia. DPIRD believes there is merit in adopting a zoned approach which considers the cumulative effect of the combined farming effort and supports the sharing of resources between farmers.

Figure 1.1 Proposed layout of the SCADZ in Albany, including the pre-existing leases and proposed extent of the new areas

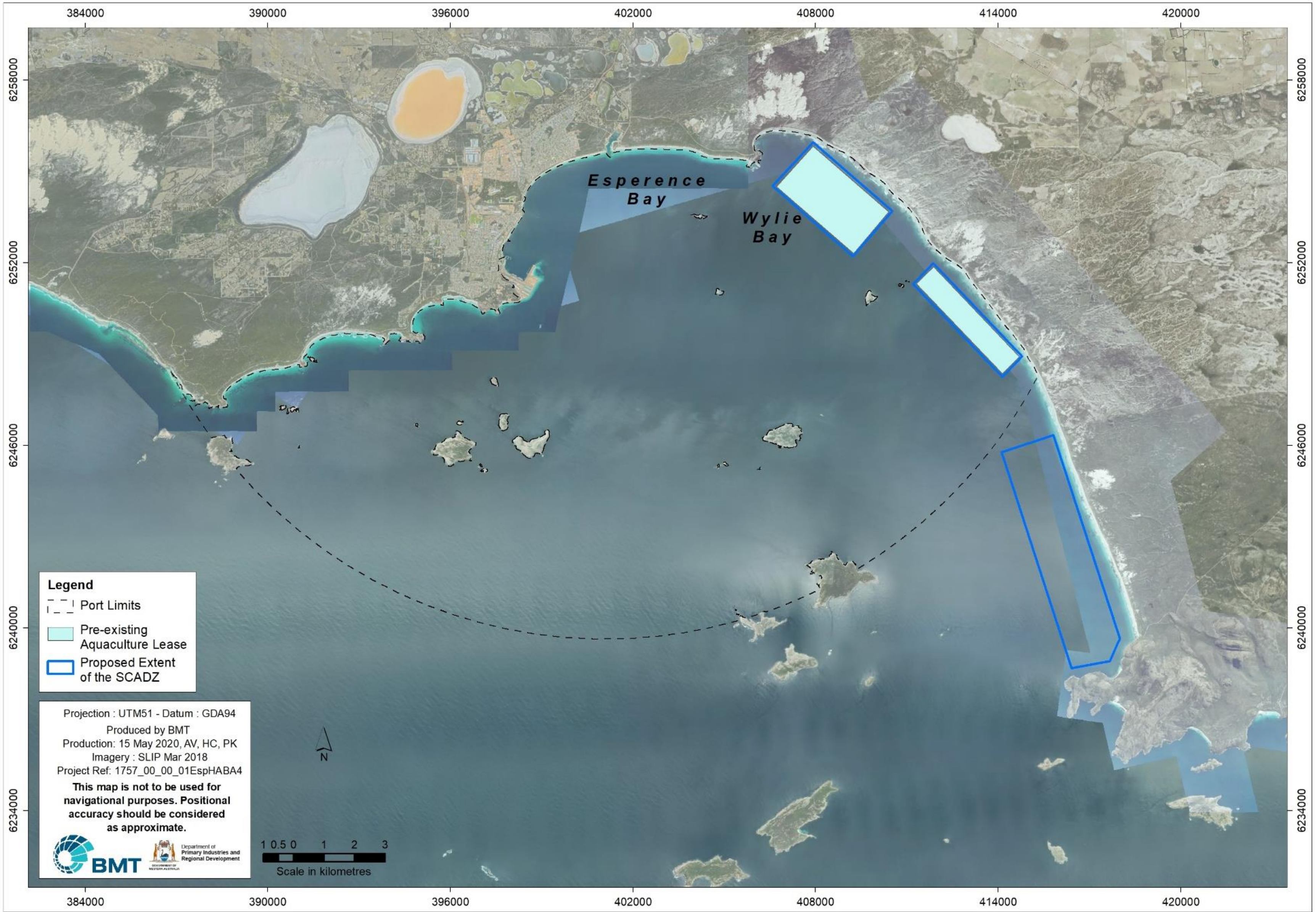


Figure 1.2 Proposed layout of the SCADZ in Esperance, including the pre-existing leases and proposed extent of the new areas

1.2 Purpose of this document

To support the development of the SCADZ, DPIRD engaged BMT to conduct a desktop assessment of the potential impacts of the proposed farming operations on the local marine environment. Risks were assessed in the context of EPA (2016a, b, c, d, e, EPA 2018) and DoH (2016, 2017). The specific objectives of the assessment are outlined in Table 1.1.

Table 1.1 Assessment objectives

Objective	Section / Appendix
Conduct a review of relevant regional studies undertaken to date	Section 3; All Sections
Determine the potential losses of benthic communities and habitats due to placement of infrastructure, as per EPA guidance (EPA 2016a, b)	Section 4
Estimate the carrying capacity of the system, based on local food availability	Section 5
Determine the potential for sediment nutrient enrichment beneath the existing and proposed grow-out infrastructure	Section 6
Review the risks posed by harmful algal blooms (HABs) within the context of the Western Australian Shellfish Quality Assurance Programme (WASQAP)	Section 7

At the request of DPIRD, the report focuses primarily on the Albany marine environment. The only exception is the assessment of potential losses of benthic communities and habitats, which were examined in the broader context of both the Albany and Esperance marine environments.

2 Proposed Operations

2.1 Implementation

The proposed Albany component of the SCADZ will be comprised of several leases at Oyster Harbour (OH), one in Frenchman Bay, two at Mistaken Island and a new lease in south-eastern Princess Royal Harbour (PRH) (Figure 1.1). Details of the SCADZ proposal are listed in Table 2.1.

Table 2.1 Project details

Component	Description
Proponent	Department of Primary Industry and Regional Development (DPIRD)
Indicative species	To be confirmed (currently *Saccostrea glomerata*, *Pinctada imbricata fucata* and *Mytilus galloprovincialis*)
Location	Albany (leases distributed across Oyster Harbour, Frenchman Bay, Princess Royal Harbour and waters adjacent to Mistaken Island).
Area	~803 ha (based on the proposed full extent)
Infrastructure	• To be confirmed (currently a combination of inter-tidal and sub-tidal culture systems). • Inter-tidal triplicate lines of oyster baskets secured using strainer and riser posts or lines of floating baskets secured with anchors • Sub-tidal lines of floating oyster baskets • Sub-tidal mussel lines, consisting of horizontal long line secured with anchors, with attached vertical culture ropes. • Infrastructure will be sited to avoid seagrass beds.
Ancillary	Small trailer-able vessels
Broodstock	Spat sourced from the Albany Shellfish Hatchery or other suitable hatchery
Construction	To be confirmed (e.g. inter-tidal strainer posts can be driven into the sandy bottom by jetting water from a fire pump to form holes for the posts; sub-tidal anchors can be screwed into the sandy substrate at either end of the horizontal longlines)
Operation details	To be confirmed
Schedule	To be confirmed

2.2 Indicative species

Shellfish farming has been conducted in Albany since 1991 when Ocean Foods International first established in OH. Shellfish farmed in Albany include *Saccostrea glomerata* (Sydney rock oyster, grown in OH), *Pinctada imbricata fucata* (Akoya pearl oyster, grown in King George Sound [KGS]) and *Mytilus galloprovincialis* (blue mussel, grown primarily in KGS and OH). The expectation is that these species will continue to be farmed, though other species may also be grown in the future if found suitable and profitable. The current and proposed leases are shown in Figure 1.1.

S. glomerata is an iconic Australian seafood product that attracts a premium in Australian seafood markets (Figure 2.1). Native to the east coast of Australia, *S. glomerata's* natural habitat is the inter-tidal and shallow sub-tidal zones. This species attaches to substrate by means of an excreted organic adhesive that forms calcareous cement over time (Harper 1996).

S. glomerata are filter-feeders and net extractors from the water; the main component of food being phytoplankton, oocytes, bacteria, and small organic particles suspended in the water column (ACWA 2013). *S. glomerata* tolerates a wide range of water temperatures, salinity and pH (O'Connor and Dove 2009), but is metabolically depressed at temperatures below 12°C (Jonathan Bilton, Manager, Albany Shellfish Hatchery pers. comm).

M. galloprovincialis is native to Australia and naturally distributed in cooler waters along the southern coast of Australia (Figure 2.1). *M. galloprovincialis* is the only commercial species of mussel sold in quantity in Australia and is grown in clean, sheltered waters at depths between 5 and 20 m. The species range extends from Eden, New South Wales to southern Western Australia. Collected and grown on droplines, they loosely attach to the lines using silky fibres called byssus threads, allowing them to drift to maximise water intake for feeding. Similar to *S. glomerata*, *M. galloprovincialis* is a filter-feeding organism that can tolerate a wide range of water temperatures, salinity and pH.

P. fucata is also native to and farmed in Australia, but rather to produce pearls (Figure 2.1). *P. fucata* is a shallow water filter-feeder with a wide distribution, straddling temperate and tropical latitudes. It tolerates water temperatures between 14–26°C and grows best in salinities between 32–35 ppt (O'Connor, Lawler and Heasman 2003).

Mytilus galloprovincialis

Saccostrea glomerata

Pinctada imbricata fucata

Figure 2.1 Indicative shellfish species proposed to be cultured

2.3 Indicative culture methods

The culture methods described here are indicative and based on knowledge of best-practice methods currently employed in the Albany / Esperance regions. These methods may change with advancement in technologies and processes.

2.3.1 Sydney Rock Oysters

Oyster spat will be sourced from the Albany Shellfish Hatchery (or similar suitable hatchery if necessary) at an age of ~100 days, before being transferred to baskets (with a fine mesh insert) secured to longlines. The longlines will vary in length depending on the size of the lease. Each longline will be spaced approximately 10 m apart, allowing for ease of access for maintenance and harvesting.

Typical sub-tidal longlines consist of a series of submerged baskets attached to an extended rope (the “longline”) (Figure 2.2). The longlines are secured with anchors placed at either end of the longline. Longlines may be installed as either inter-tidal or sub-tidal systems (Figure 2.2). The inter-tidal and sub-tidal systems in OH employ a flip-basket design, whereby the baskets can be ‘over-turned’ periodically to expose the shellfish to air. The approach helps to maintain shellfish condition and control biofouling.

Source: BMT

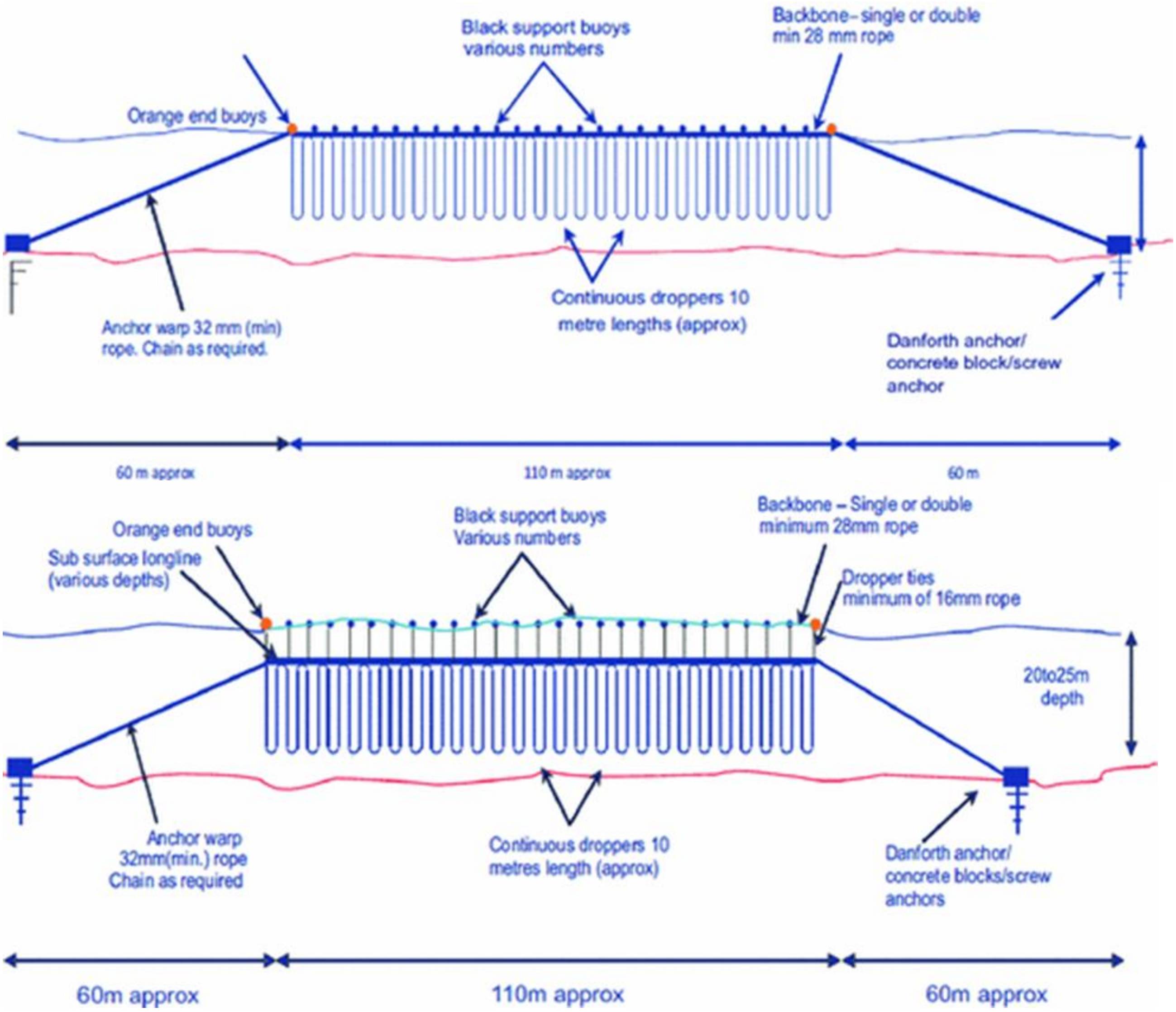
Figure 2.2 Inter-tidal longline and sub-tidal floating longline system with baskets from Oyster Harbour

As oysters are filter-feeders, their culture relies entirely on natural food sources. Interventions during growth may include grading at an offshore facility, movement of shellfish between sites and drying (if using floating upweller systems or FLUPSYs for example). Grading involves the removal of oyster baskets from longlines, removal of biofouling, manual sorting of oysters based on size into fresh baskets, followed by their re-deployment to new baskets. The type of interventions employed is at grower discretion, and may change with advancement in culture technologies and processes.

Harvest occurs when they reach ~2–3 years of age or an average size of ~50 mm. During harvest, baskets and other infrastructure are removed from the longlines and the oysters harvested. Oysters destined for local sale are visually sorted and blasted with fresh water; oysters for wholesale are sorted visually, barnacles chipped off by hand and bagged for transport. Any biofouling is returned to open water. Disposal of biofouling to sensitive environments, such as seagrasses, is avoided as per best practice guidelines.

2.3.2 Blue mussels and Akoya pearl oysters

M. galloprovincialis and *P. fucata* are grown in KGS using a combined floating longline and dropline system. Longlines of 100–180 m length are hung across the surface of the water, with floats on either end. The longlines are secured with heavy anchors. Along each longline are a series of droplines, to which the shellfish attach directly using their byssal thread (Figure 2.3).



Source: Goseberg et al. 2017

Figure 2.3 Cross-section of a surface (upper panel) and subsurface longline (lower panel) mussel farm

Harvest occurs at ~9-12 months of age when shellfish achieve a marketable size of ~50 mm. The droplines are constantly maintained to remove size shellfish and ensure an appropriate density of shellfish. This prevents the droplines from becoming overweighted, and maintains the productivity of the system.

2.3.3 Abalone

Green-lip abalone (*Haliotis laevigata*) have been farmed in Wylie Bay and Esperance Bay since 2017 using artificial reefs, known as ABITATS (Figure 2.4). The rearing process, known as sea-ranching, involves placing hatchery reared juveniles on the artificial reefs where they are left to grow to marketable size.

As well as the two existing leases in Esperance, a new area will be established off Cape Le Grand (see Figure 1.2). This will substantially increase the area available for the operators to install and monitor further artificial reefs. It is expected that this species will continue to be farmed, though other species may also be grown in the future if found to be suitable and profitable.

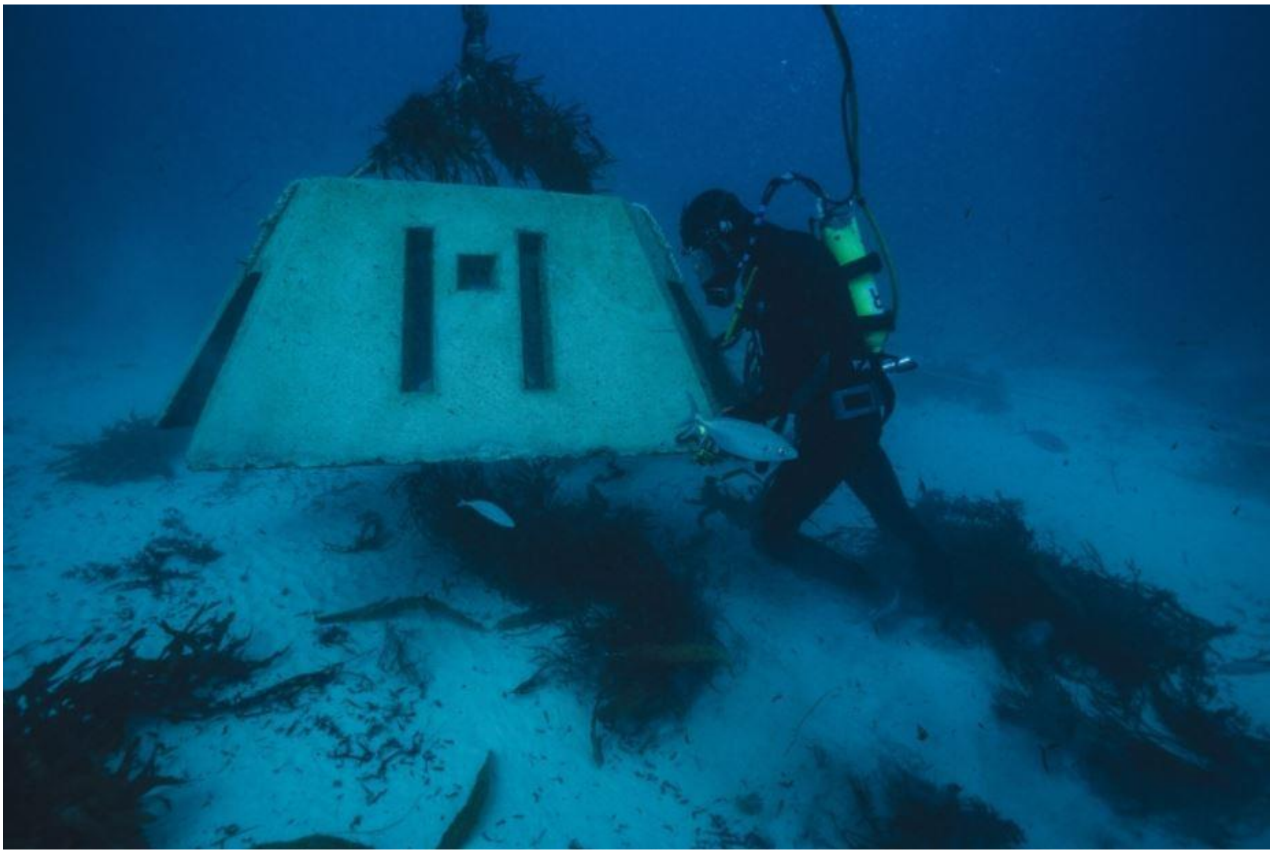


Figure 2.4 Typical ABITAT layout on the substrate

2.4 Regulatory framework

2.4.1 Environmental approvals

Shellfish farming is ostensibly undertaken in shallow waters using purpose-built structures, incorporating vertical (strainer and riser) posts, horizontally suspended baskets and/or rope lines suspended vertically. Unlike finfish farming, shellfish farming does not require inputs of supplementary feeds, and instead relies on natural food sources (phytoplankton or drift algae). In the absence of artificial feeding, the environmental risks associated with shellfish culture are considered relatively benign (Bulmer et al. 2012; Dumbalk and McCoy 2015).

DPIRD considers that the potential impacts of farming can be managed and regulated through aquaculture licences and leases and via the application of Management and Environmental Monitoring Plans, which are required under the *Fisheries Resources Management Act 1994*.

The Environmental Protection Authority (EPA) considers this a reasonable position and acknowledges that DPIRD's approach to quantify the cumulative impacts of farming and develop management strategies for the entire zone will lead to better environmental outcomes (EPA 2020).

EPA is supportive of DPIRD committing to a robust desktop assessment (this report) to ensure the potential cumulative effects of the SCADZ are identified, well understood and manageable.

2.4.2 Department of Health / WASQAP Approvals

The culture of shellfish for human consumption is regulated by the Department of Health, pursuant to the criteria listed in the Western Australian Shellfish Quality Assurance Program (WASQAP) Operations Manual 2017 (DoH 2017) and the Australian Shellfish Quality Assurance Program (ASQAP) Operations Manual 2019 (ASQAP 2019). Under WASQAP, shellfish include but are not limited to, edible oysters, cockles, clams, uneviscerated scallops and mussels.

Filter-feeding bivalves consume and filter large quantities of water and are therefore susceptible to the bio-accumulation of bacteria, viruses, toxins, heavy metals, chemicals and other harmful substances. The consumption of contaminated shellfish may lead to foodborne illnesses.

Shellfish farmers must obtain a classification for the shellfish lease in accordance with the WASQAP (2017) and the ASQAP (2019) Operations Manuals. Classification must be undertaken prior to the commercial harvesting and supply of product for human consumption. The classification of shellfish leases is assigned following a comprehensive assessment of potential pollution sources as per the Australian and New Zealand Food Standards Code.

The risks posed by harmful algal blooms (HABs) in OH and KGS are known to operators and the Department of Health, and regulated under WASQAP. The risk assessment therefore focussed on PRH, which is yet to be classified. Risks in PRH were determined based on the potential for operations to effect changes to regional water quality, and particularly the characteristics likely to increase the probability of an algal bloom.

3 Environmental Significance

3.1 Physical environment

Water circulation in the Albany region, especially in Princess Royal Harbour (PRH) and King George Sound (KGS) is primarily wind driven. Tides are relatively weak with a neap spring tide of 1.1 m. Wind driven circulation in KGS is predominantly anti-clockwise in summer and clockwise in winter (Mills and Brady 1985, GEMS 2007).

PRH is well protected from the otherwise highly energetic wave climate of the south-west (McClatchie et al. 2006). Mills and D'Adamo (1993) for example found that up to 30 million m³ of water transits to and from PRH within 8 hours of rising tides and 16 hours of falling tides. Water passes through the entrance of the Harbour at current speeds of up to 0.5 m/s.

KGS is primarily a salt-water ecosystem with minimal to no influence from freshwater sources. Oyster Harbour (OH) and PRH however display characteristics typical of estuarine ecosystems. OH in particular is characterised by seasonal fluctuations in salinity due to riverine inputs, which contribute to stratification in both salinity and temperature (Thomson 2018). PRH is generally well mixed but occasionally experiences weak stratification (Ecologia 2009).

Sediment sampling was conducted in PRH and KGS in 2014 prior to the most recent maintenance dredging in 2015 (BMT Oceanica 2014). Metal concentrations were below screening levels set by the NAGD, and sediment nutrient values for total nitrogen and total phosphorus were within the range expected for estuarine and coastal waters in the region (BMT Oceanica 2014).

3.2 Water quality

Historically, both OH and PRH have experienced significant anthropogenic nutrient enrichment due to riverine (seasonal) and/or urban and agricultural (year-round via drains) runoff. This has resulted in algal blooms, as well as substantial epiphytic and macroalgal growth on seagrass and subsequent declines in seagrass meadows which previously dominated both estuaries (Brearley 2005).

Typical healthy oligotrophic systems in the south west of Western Australia maintain phytoplankton concentrations (as indicated by chlorophyll-a) at around 1 µg.chlorophyll-a/L (Brearley 2005); whereas, according to ANZECC/ARMCANZ (2000), enriched ecosystems exceed 3 µg.chlorophyll-a/L. Historical data collected between 1988 and 2018 point to significant differences in the level of productivity between the OH, KGS and PRH water bodies. While KGS is clearly oligotrophic (0.05 and 1 µg.chlorophyll-a/L), OH and historically PRH regularly approach/ed or exceed/ed the ANZECC/ARMCANZ (2000) trigger for eutrophic systems, respectively); this is particularly the case for OH which has recorded concentrations as high as 11 µg.chlorophyll-a/L (Hillman 1991, Thomson 2018).

3.3 Ecology

3.3.1 Fauna

The marine flora and fauna of Albany are primarily temperate species with a small proportion of endemic species (Wells 1988). Marine fauna of conservation significance include humpback whales (*Megaptera novaeangliae*) and southern right whales (*Eubalaena australis*), common dolphins (*Delphinus delphis*) and bottlenose dolphins (*Tursiops truncatus*) as well as Australian sea lions (*Neophoca cinerea*) and New Zealand fur seals (*Arctocephalus forsteri*) (DPaW 2016). The south coast of Western Australia is also a significant resource for the conservation of seabirds and shorebirds (Commonwealth of Australia 2012).

A search of the Commonwealth Protected Matters returned a number of marine fauna of conservation significance (Table 3.1). Both the Albany and Esperance regions record substantial numbers of mammals, fish and birds that either reside, breed or transit within the area.

Table 3.1 Summary of marine fauna from the EPBC Protected Matters report

Group	Common Name	Species Name	Conservation Status
Mammals	Minke whale	*Balaenoptera acutorostrata*	
	Sei whale	*Balaenoptera borealis*	Vulnerable
	Bryde's whale	*Balaenoptera edeni*	
	Blue whale	*Balaenoptera musculus*	Endangered
	Fin whale	*Balaenoptera physalus*	Vulnerable
	Pygmy right whale	*Caperea marginata*	
	Common dolphin	*Delphinus delphis*	
	Southern right whale	*Eubalaena australis*	Endangered
	Risso's dolphin	*Grampus griseus*	
	Dusky dolphin	*Lagenoryhchus obscurus*	
	Humpback whale	*Megaptera novaeangliae*	Vulnerable
	Orca	*Orcinus orca*	
	Indo-Pacific bottlenose dolphin	*Tursiops aduncus*	
	Bottlenose dolphin	*Tursiops truncates*	
	New Zealand fur seal	*Arctocephalus forsteri*	
	Australian sea lion	*Neophoca cinerea*	Vulnerable
Reptiles	Loggerhead turtle	*Caretta caretta*	Endangered
	Green turtle	*Chelonia mydas*	Vulnerable
	Leatherback turtle	*Dermochelys coriacea*	Endangered
Fish	Grey Nurse sharks	*Carcharias taurus*	Vulnerable
	White shark	*Carcharodon carcharias*	Vulnerable
	Whale shark	*Rhincodon typus*	Vulnerable
	Southern pygmy pipehorse	*Idiotropiscis australe*	
	Gales pipefish	*Campichthys galei*	
	Upside-down pipefish	*Heraldia nocturna*	
	Short-head seahorse	*Hippocampus breviceps*	
	Rhino pipefish	*Histiogamphelus cristatus*	
	Brushtail pipefish	*Leptoichthys fistularius*	
	Smooth pipefish	*Lissocampus cordalis*	
	Javelin pipefish	*Lissocampus runa*	
	Sawtooth pipefish	*Maroubra perserrata*	
	Bonyhead pipefish	*Nanocampus subosseus*	
	Red pipefish	*Notiocampus ruber*	
	Leafy seadragon	*Phycodurus eques*	
	Common seadragon	*Phyllopteryx taeniolatus*	
	Pugnose pipefish	*Pugnaso curtirostris*	
	Gunther's pipehorse	*Solegnathus lettiensis*	
	Spotted pipefish	*Stigmatopora argus*	
	Widebody pipefish	*Stigmatopora nigra*	
	Hairy pipefish	*Urocampus carinirostris*	
	Mother-of-pearl pipefish	*Vanacampus margaritifera*	
	Port Phillip pipefish	*Vanacampus phillipi*	
	Long-snout pipefish	*Vanacampus poecilolaemus*	

3.3.2 Habitats

The Albany region supports an abundance of benthic primary producer habitats, particularly seagrasses. Seagrasses support populations of juvenile fish and benthic invertebrates, while the shallow nearshore regions of PRH and OH are important feeding areas for water birds (Kirkman 1997, McClatchie et al. 2006, Strategen 2008). Seagrass habitats provide refuges for juvenile fish including commercially / recreationally fished species such as pink snapper (*Pagrus auratus*) and benthic invertebrates (Kirkman 1997, Strategen 2008) and fulfil an important role in enhancing nutrient cycling (Kilminster et al. 2018).

Benthic habitat mapping has revealed multiple seagrass species throughout the region. In OH, *Posidonia australis* and *P. sinuosa* predominate in waters <5 metres deep. These meadows suffered extensive losses during the 1980s primarily due to nutrient and sediment inputs from the King and Kalgan rivers (Cambridge, Bastyan and Walker 2002). Seagrasses have since recovered, mainly due to extensive seagrass transplantation (Bastyan and Cambridge 2008). The meadows in PRH support a mixture of *P. australis*, *P. sinuosa* and *Amphibolis griffithii*. As with OH, these meadows have also suffered significant losses since the 1950s due to smothering from macroalgae (Bastyan 1996). Some of these losses have since been mitigated via seagrass transplantation and the implementation of a Seagrass Rehabilitation and Monitoring Management Plan (Oceanica 2011). In KGS, seagrass meadows of *P. coriacea* dominate the nearshore regions along Middleton Beach (BMT Western Australia 2018), while meadows of *Posidonia* spp. and *Amphibolis* spp. occur in various densities along the shoreline in the south-west.

While not as prevalent as seagrass communities, macroalgal habitats fulfil an important ecological niche in the Albany region. Changes in nutrient loads have under some circumstances resulted in smothering and subsequent dieback of seagrass due to the macroalga *Cladophora sp.* (Hillman 1991).

Natural reefs consisting of *Ostrea angasi* were once a predominant feature of OH, but with the expansion of urban development most of the reefs have now disappeared. Recent efforts by the Nature Conservancy in conjunction with other organisations have been made to restore these reefs with some success (The Nature Conservancy 2020). The current aim is to re-establish approximately 800 m^2 of oyster reefs.

Sand habitats occupy the spaces between the seagrasses and macroalgal beds. These sand habitats are particularly common in the deep waters of KGS.

3.3.3 Introduced marine species

Introduced marine species (IMS) are those introduced by human activities such as shipping (CA 2013a). IMS have the potential to significantly impact marine industries and the environment. Sixty species are known to have been introduced into Western Australia where they are now established; most being cool water temperate species (Wells et al. 2009). DPIRD has recorded 25 IMS in Albany (Ecologia 2007, APA 2013, DoF 2016), including:

- Pacific oyster (*Crassostrea gigas*);
- Toxic dinoflagellate (*Gymnodinium catenatum*);
- Ascidian tunicate (*Ascidiella aspersa*);
- Three species of bryozoans (*Cryptosula pallasiana*, *Bugula flabellata*, and *Bugula neritina*);
- European fan worm (*Sabella spallanzanii*);
- Codium (*Codium fragile spp*.).

3.4 Social environment

OH is listed as a nationally significant wetland and is highly valued in the Albany community. KGS, PRH and OH are all areas of historical importance for indigenous and non-indigenous culture and heritage. The harbours and rivers which flow into KGS were the focus of traditional Aboriginal domestic life and are important areas of traditional mythological significance (Ecologia 2009).

In addition, there are several shipwrecks throughout this region which are of significant maritime heritage. Other social uses of the area include commercial and recreational fisheries and tourism.

As a component of the SCADZ implementation, DPIRD contracted South Coast NRM to conduct a community consultation report in 2019. Two workshops, an online survey and direct contact with stakeholders were carried out to fulfil this objective.

4 Benthic Communities and Habitats

4.1 Overview of studies

Benthic communities and habitats (BCH) refer primarily to benthic-dwelling primary producing habitats and the communities they support. Geographically relevant examples include seagrasses, macroalgae and microphytobenthos.

In Albany, the distribution of BCH has been the subject of multiple studies, most of which were commissioned to support the EIA processes for non-aquaculture related proposals (Bastyan 1986, Cambridge and Walker 2002, Ecologia 2007, Bastyan and Cambridge 2008, BMT 2018, MScience 2019). The exception to this is MScience (2019), which maps the distribution of BCH across six candidate sites originally proposed for the SCADZ (MScience 2019).

Maps were produced using a composite of Sentinel-2 satellite images, which were analysed for a broad range of benthic categories (sparse seagrass, seagrass, bright sand, dark sand and sand where it could not be differentiated). The composite images were ground-truthed using imagery captured in situ. Seagrass and sparse seagrass accounted for 74% of the BCH recorded (Table 4.1).

Table 4.1 Distribution of benthic habitat classes within candidate sites

Site	Area (ha[1])	Habitat class				
		Sparse seagrass	Seagrass	Sand	Bright sand	Dark sand
Albany Sector						
Shoal Bay	149.4	7%	89%	3%	0%	0%
Mistaken Island	150.3	28%	63%	6%	3%	0%
Frenchman Bay	109.6	11%	89%	0%	0%	0%
Gull Rock	80	30%	9%	43%	17%	0%
Oyster Harbour	535.2	0%	51%	0%	6%	44%
Esperance Sector						
Cape Le Grand	1346.8	31%	43%	26%	0%	1%

Source: MScience (2019)
Notes:
1. ha = hectares

Data on the distribution of BCH in Esperance is scarce and limited to the results of two studies (DPaW 2006, BMT 2016). BMT (2016) used satellite and aerial imagery combined with data from a towed video survey to produce a detailed map showing the distribution of non-vegetated and vegetated reef, with the latter encompassing categories for seagrass, macroalgae, sand and reef. Further studies completed by the Department of Parks and Wildlife surveyed all habitats in the Recherche Archipelago, to produce a complementary map of the nearshore regions (DPaW 2006).

4.2 Potential impacts

The potential for shellfish farming to impact BCHs is generally considered low, and where unavoidable, restricted to the direct effects of 'piercing' or 'smothering' due to the placement of posts, anchors and artificial substrates. The potential for indirect effects due to shading or benthic nutrient enrichment is also considered very low, and fully recoverable under best-practice aquaculture protocols, which routinely include fallowing (Table 4.2).

Table 4.2 Most likely direct and indirect impacts to benthic communities and habitats

Potential impacts	Context	Likelihood of occurrence and recovery
Direct loss of benthic communities and habitat (BCH)	• Aquaculture infrastructure (posts, anchors) directly removing or obstructing benthic habitats (particularly seagrass) • Vessel anchors / props directly removing benthic habitats (particularly seagrass)	• High (irrecoverable) • Low (recoverable)
Secondary and tertiary loss of BCH (shading / smothering)	• Shading of benthic habitats from floating baskets, longlines and artificial reefs • Organic enrichment in the sediments due to the bio-deposition of shellfish wastes	• Low (recoverable) • Low (recoverable)

4.3 Methods

4.3.1 Local assessment units

EPA (2016a) provides a risk-based spatial assessment framework for evaluating cumulative loss or serious damage to BCH. The framework has been applied here to determine the extent of direct and/or indirect impacts to seagrass and macroalgal habitats due to construction and operational activities. The assessment consider all losses (a) prior to human-induced disturbance, (b) existing at the time of the proposal and (c) remaining after implementation of the proposal.

LAUs are defined areas where the impact of a proposal on BCH is spatially assessed (EPA 2016b). LAUs are not standardised, and must be defined individually for each proposal, although as a guide a LAU in the order of 50 km^2 (5000 ha) should be used for assessments in WA (EPA 2016a).

The SCADZ project is unique in that its footprint is non-contiguous, i.e. it has several components each spread across different geographic areas, and each with differing ecologies. For this reason, BCH losses for the SCADZ project were assessed within individual LAUs, encompassing a total area of 232 km^2.

Several historical LAUs exist within or near the proposed SCADZ: an LAU of approximately 75 km^2 in Esperance (EPA 2000) and three separate LAUs in Albany, each of differing size (Figure 6.18 in Ecologia 2007). The Esperance LAU does not extend to the boundaries of the current and proposed SCADZ leases, so a new LAU of 100 km^2 was created (Figure 4.4) to cover the current and proposed leases as well as the surrounding nearshore waters.

While there are three existing LAUs in the Albany region: one in Princess Royal Harbour (PRH; LAU 1, 38 km^2) and two for King George Sound (KGS; LAU 2 and 3, 74 and 64 km^2, respectively); none extend to Oyster Harbour (OH). For the Albany component of the SCADZ, BCH loss was assessed in each of four separate LAUs, consisting of LAU 1, 2 and 4, with 4 being a newly established LAU for OH (Figure 4.3; 20km^2). This approach was approved through direct consultation with the EPA.

4.3.2 Habitat mapping

Habitat maps for the present-day scenario were created using a combination of publicly available and custodian data together with high resolution aerial / satellite imagery, which provided coverage for the entire area of interest:

- Bastyan (2015) Benthic habitat data for Middleton Beach, King George Sound - produced for the City of Albany;
- BMT Oceanica (2016). Benthic habitat data for Wylie Bay - produced for Ocean Grown Abalone;
- DoW (2013a, b, c). Oyster Harbour Seagrass 1988-2006;
- DPaW (2006). Marine Habitats of Western Australia;

- DWER (2021). Seagrasses in four estuaries in Western Australia's South West 2017-20;
- Ecologia (2007). Benthic habitat data for King George Sound and Princess Royal Harbour - produced for the Albany Port Authority;
- Oceanica (2006). Albany Waterfront Project - Protected Harbour Development Princess Royal Harbour Benthic Habitat Mapping – produced for Landcorp;
- Kirkman (1997). Seagrasses of Australia.

The baseline maps were created as per EPA guidance (EPA 2016a, b). This involved replacing areas of known direct loss – such as from dredging, land reclamation, marina and port developments – with either the original habitat or, if that information was not available, a suitable nearby habitat. In areas where the losses resulted from indirect impacts, estimates were based on the best historical information available.

In the case of OH, areas of known loss (i.e. boating channel, Emu Point boat harbour) were assumed to historically be 100% seagrass. Where the historical coverage was ambiguous or unknown, a conservative approach was taken which again assumed the Harbour was 100% seagrass to a depth of 6 m[1].

For PRH, there existed a precedent for BCH loss calculations (Ecologia 2007) which assumed the Harbour was seagrass dominated prior to European settlement. We adopted a similar approach by assuming all waters below the high tide mark (including the area reclaimed by the Port of Albany) were once covered in seagrass, including the Princess Royal Sailing Club, Albany Waterfront Marina and the Port of Albany.

Current day scenarios were used in KGS and Esperance, reflecting the fact none of these regions have experienced major anthropogenic impacts on benthic habitats historically or in recent times.

Estimates of future losses due to shellfish farming activities were calculated based on the areas impacted by development and operational activities. Habitats were broadly categorised as seagrass, macroalgae and sand. The broad categorisation was used to best represent the variety of data sources used, and the differing level of resolution applied historically.

All mapping and area calculations were completed using ArcGIS. Bathymetry estimates were derived in Global Mapper. The approach was based on several assumptions as outlined in Table 4.3.

[1] 6 m was used as a boundary as it represents the theoretical upper depth limit for the seagrass species found in OH.

Table 4.3 Assumptions underpinning habitat loss assessment

Assumption
• Seagrass habitats identified in the ground-truthing match the aerial / satellite imagery, i.e. some seagrass habitats may have been covered with macroalgae (such as the genus *Cladophora*), but differentiation between the two is not possible using aerial / satellite imagery
• The depth limits used in the baseline scenario accurately reflect the limit for dense, healthy meadows. Some ephemeral meadows may have grown beyond this limit pre-European settlement. • The assessment was limited to areas where dense, healthy, permanent meadows were assumed to have grown.
• A conservative assessment was used by assuming the infrastructure (longlines, riser posts etc.) covered the entire lease. This is unlikely to occur.
• The previously mapped areas are representative of present day habitats.
• Macroalgal coverage was likely underestimated, as previous studies focused on seagrasses.
• Flat oyster (*Ostrea angasi*) reefs were not included in the loss assessment as their historical distribution in Oyster Harbour is not known for individual reefs (Peter Cook pers. comm). Presently, no substantial areas of reef exist in Oyster Harbour, except for those which have been rehabilitated by the Nature Conservancy. The construction of these reefs covers an approximate area of 800m^2.
• ABITAT placements in Esperance will be confined to non-vegetated habitats only
• Esperance leases will be used solely for abalone ranching

4.3.3 Loss assessment calculations

Historical loss assessments were based on the precedents set for the Albany Port Authority (now Southern Ports Authority) and City of Albany. Historical losses in Esperance were assumed to be zero, given the lack of evidence for anthropogenically driven changes to BCH.

Losses to BCH due to proposed shellfish farming activities were considered in the context of EPA (2016b) (Table 4.4) and the cause-effect pathways shown in Figure 4.1 and Figure 4.2. The approach considers the potential for irrecoverable and recoverable losses due to the construction and/or the operation of the farms, and the effect of differing types of farming infrastructure (inter-tidal versus sub-tidal). Losses were based on infrastructure consisting of 10 longlines, each of 100 m length (occupying a 1 ha area). Table 4.5 outlines the dimensions of the infrastructure, based on SEAPA's 25 L basket size configuration, one of the more widespread set-ups used by aquaculture operators currently.

Table 4.4 Zones of impact and influence

Zone of impact	Criteria	Contributing factor
High	Irrecoverable loss (>5 years to recover)	• Assigned to benthic communities and habitats expected to be lost directly due to placement / drilling of infrastructure i.e. from piercing effects of posts or smothering due to artificial habitats. Losses in this category are considered irrecoverable.
Moderate	Recoverable loss (<5 years to recover)	• Assigned to benthic communities and habitats affected by shading or increased sedimentation and/or nutrient enrichment. Losses in this category (if any) are considered 100% recoverable.
Influence	No material impacts	• Assigned to habitats expected to be influenced by operations, including minimal turbidity, sedimentation and nutrient enrichment.

Table 4.5 Aquaculture infrastructure dimensions used in the loss assessment

Structure type	Diameter / L* W (m)	Area (m^2)	Number per longline
Inter-tidal riser posts	0.076	0.12	32
Inter-tidal end posts	0.25	0.399	2
Sub-tidal end anchors	0.30	0.471	2
Baskets[1]	0.65 × 0.4	2.6	100

Notes:
1. Baskets = shellfish baskets designed by SEAPA (25 L size)

The percentage allocation of inter-tidal versus sub-tidal infrastructure was assumed using a combination of bathymetry and knowledge of previous use. This was an important step to correctly allocate the cause-effect pathways, which differ slightly between infrastructures (Figure 4.1 and Figure 4.2).

For the OH leases, inter-tidal infrastructures were allocated to the eastern side of the Harbour and any nearby shallow regions which fell within the inter-tidal range. Remaining parts of OH were allocated sub-tidal infrastructures.

Due to its depth, PRH was assumed to be suitable only for inter-tidal infrastructure, while all areas in KGS were assumed suitable only for sub-tidal infrastructure. While the actual configuration is likely to differ, this approach allows for a conservative estimate of potential habitat losses.

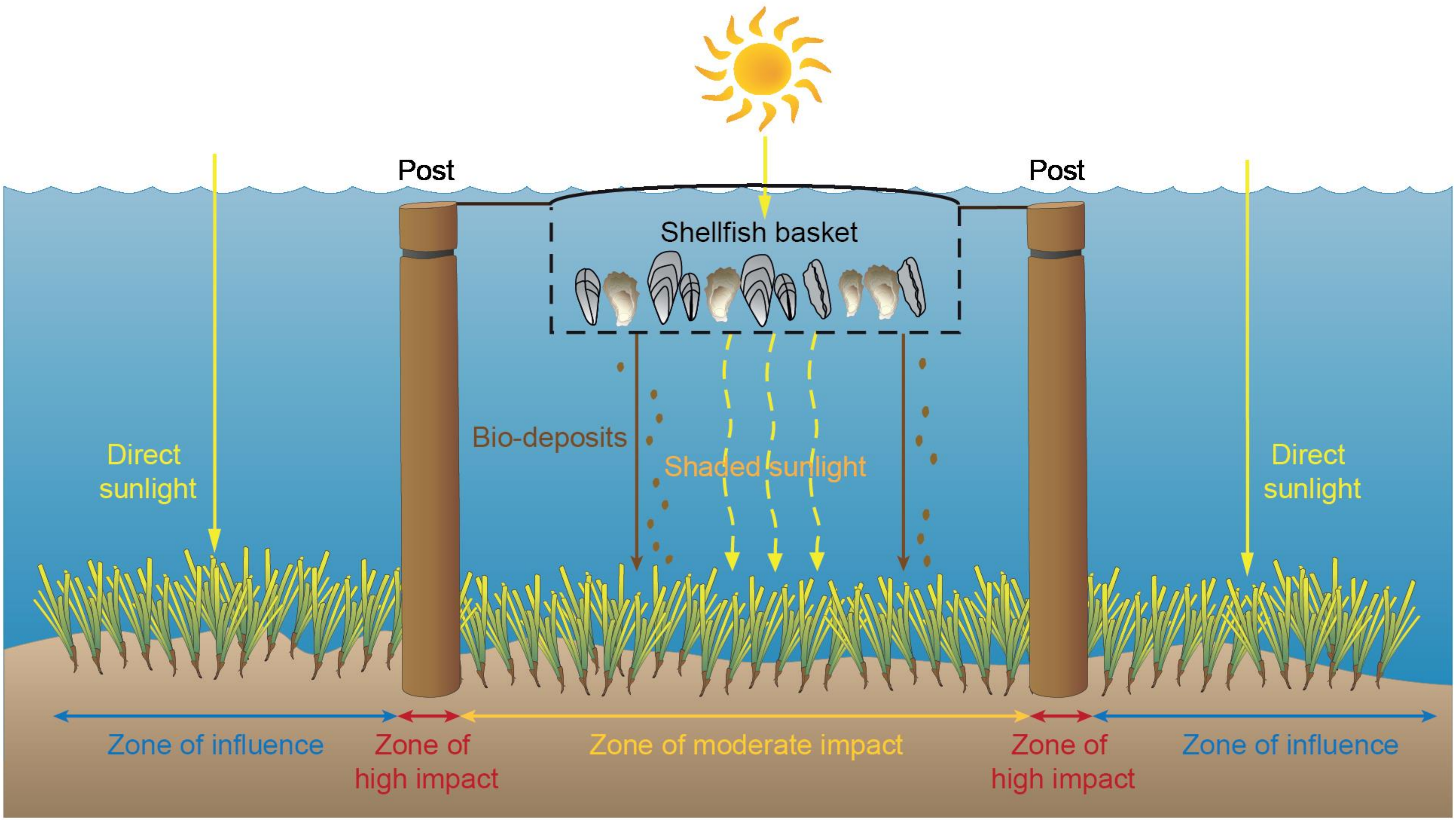


Figure 4.1 Indicative schematic of zones of impact and influence for inter-tidal shellfish aquaculture system

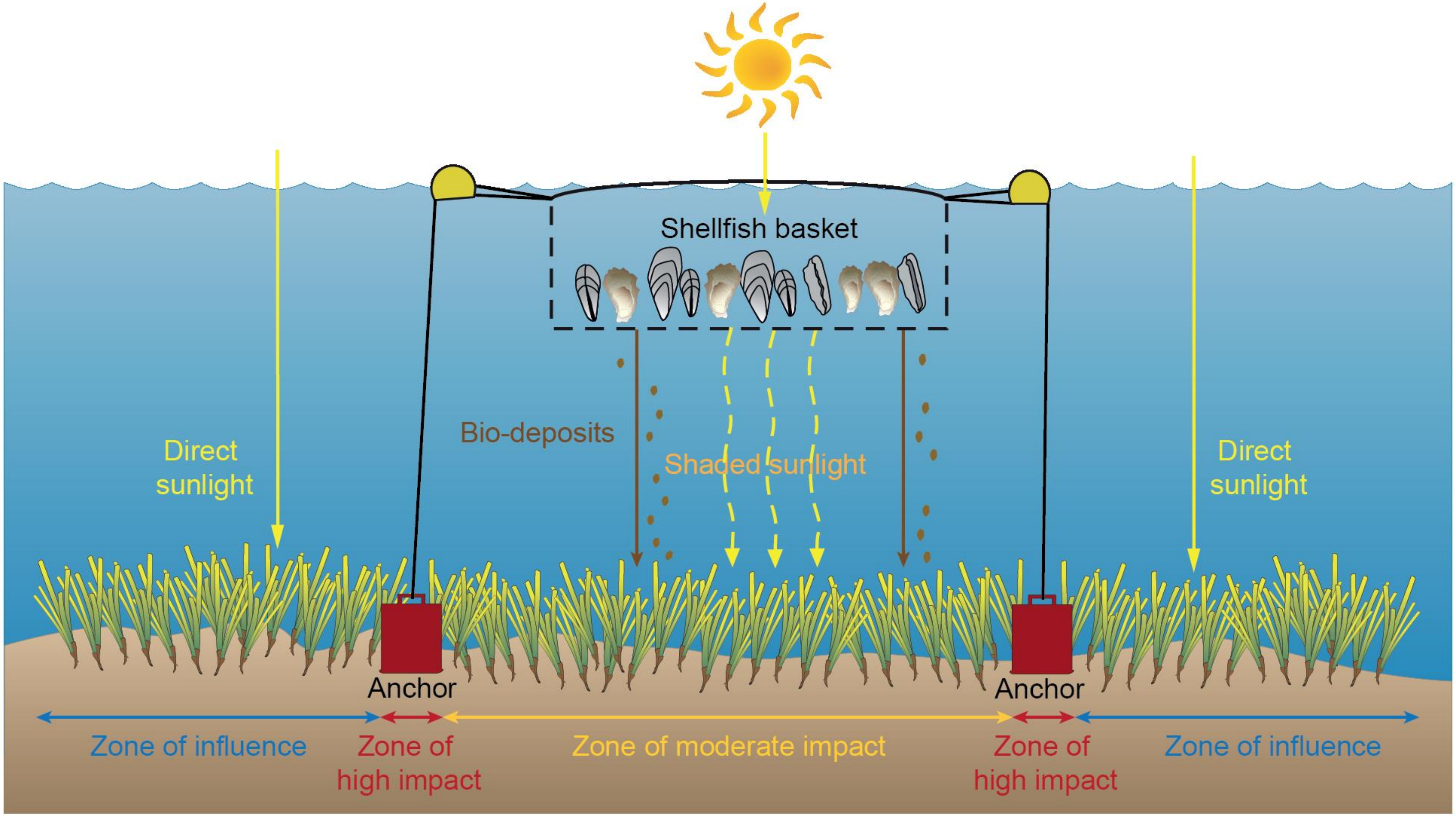


Figure 4.2 Indicative schematic of zones of impact and influence for sub-tidal shellfish aquaculture system

4.4 Results

4.4.1 Calculating historical loss

4.4.1.1 Albany

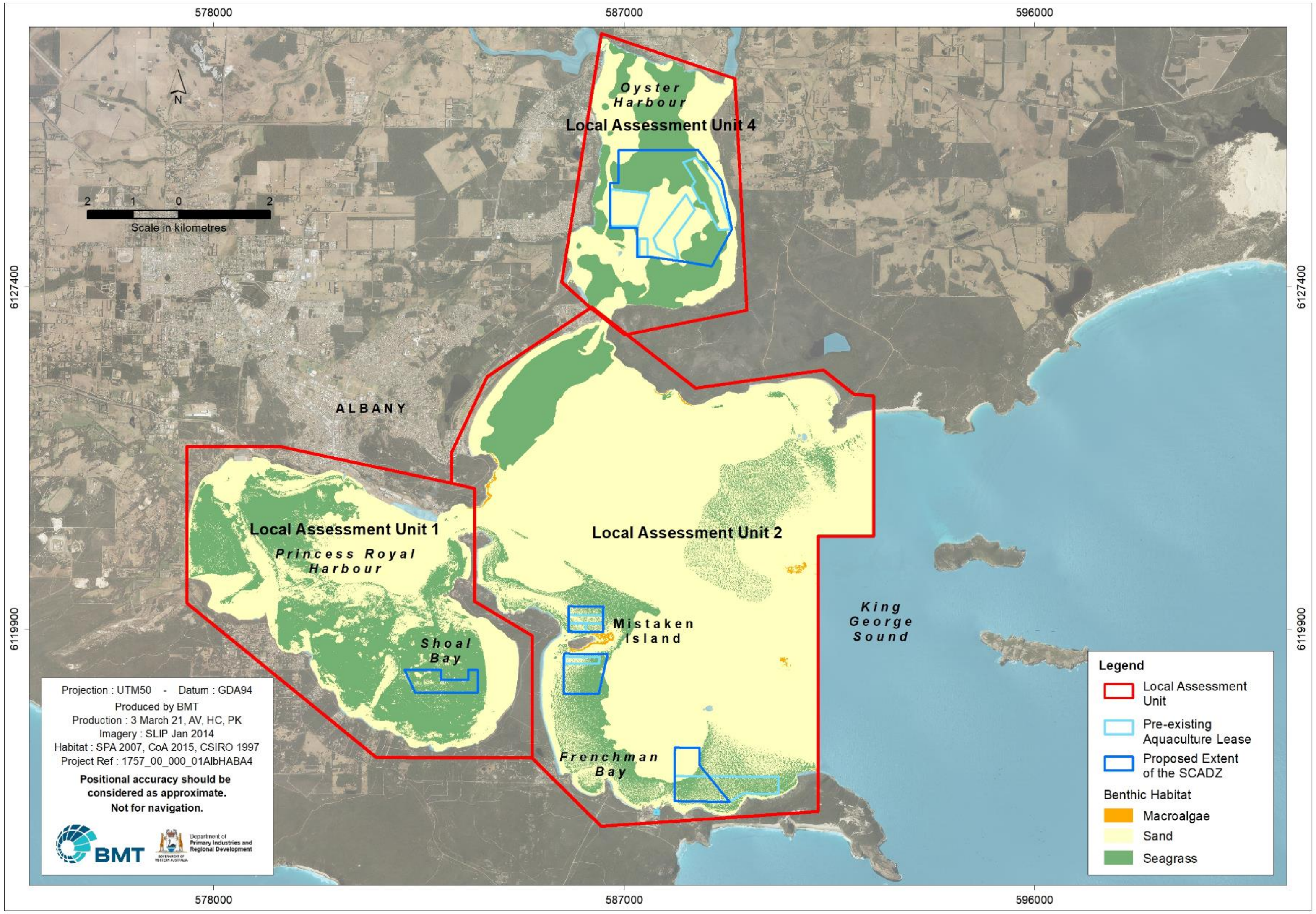
Approximately 1500 ha and 990 ha of seagrass meadows have respectively been lost in PRH and OH since European settlement.

Human use of KGS, including the dredging of the existing Albany Port Channel, has resulted in negligible seagrass losses (0.005 ha) (Ecologia 2007). For the purposes of this assessment, historical losses in KGS were therefore assumed to be zero. Table 4.6 outlines the historical and current coverage of benthic habitats in the three Albany LAUs. The contemporary spatial extent and distribution of benthic habitats is shown in Figure 4.3.

Table 4.6 Historical and current coverage of benthic communities and habitat in Albany

LAU	Bare sand (ha[1])		Seagrass (ha)		Macroalgae (ha)	
	Historical	2021	Historical	2021	Historical	2021
1 - Princess Royal Harbour	0.00	1453.90	2889.00	1385.00	0.20	0.20
2 - Inner King George Sound	5413.70	5413.70	1159.00	1159.00	21.00	21.00
4 - Oyster Harbour	52.90	908.61	1549.00	696.44	0.00	0.00

Notes:
1. ha = hectares



Notes:

1. Habitat data for Middleton Beach was sourced from City of Albany; for Princess Royal Harbour and King George Sound from the Albany Port Authority; and for Oyster Harbour from CSIRO and DWER

Figure 4.3 Current benthic habitat coverage within the three Local Assessment Units at Albany

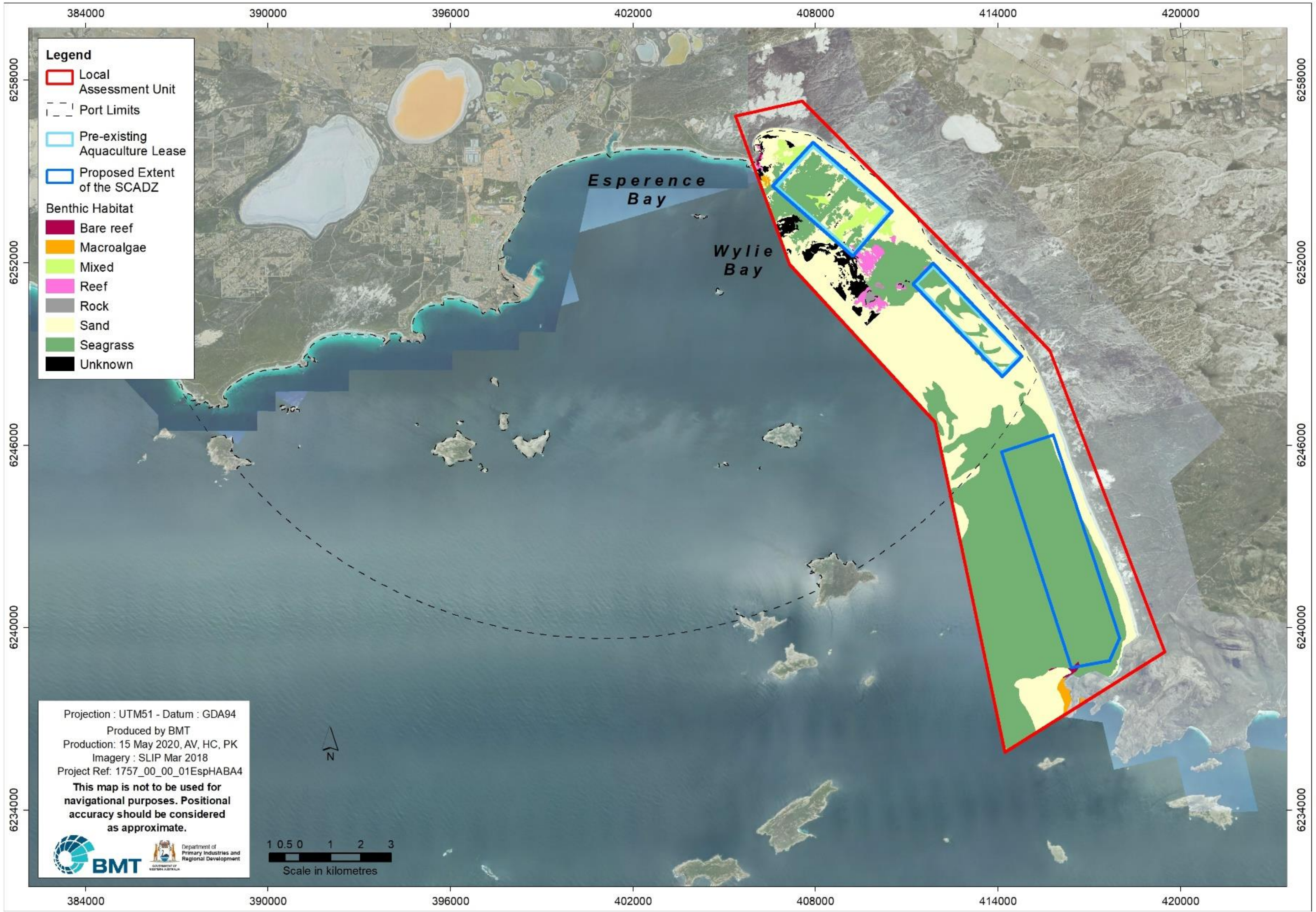
4.4.1.2 Esperance

Historical losses at Wylie Bay and Cape Le Grand have been minimal to nil as the coastline in this region has never been developed substantially, and a considerable proportion forms the Cape Le Grand National Park. As such, historical losses were assumed to be zero. Table 4.7 and Figure 4.4 outline historical and current coverage of habitat types in the Esperance LAU.

Table 4.7 Benthic communities and habitat coverage in Esperance

LAU[1]	Bare sand (ha)[2]	Rock[3] (ha)	Bare reef[4] (ha)	Coral reef (ha)	Mixed[5] (ha)	Seagrass (ha)	Macroalgae (ha)
Esperance	3501.00	13.40	12.40	99.90	161.90	4655.10	36.90

Notes:
1. historical and 2020 coverage considered the same due to no significant losses from human impacts to date
2. ha = hectares
3. Rock = granite or similar rock
4. Bare reef = bare limestone reef with no vegetation
5. Mixed = mixed assemblages of seagrass and macroalgae where neither is dominant (~50/50 coverage makeup)



Notes:

1. Habitat data was sourced from the Department of Parks and Wildlife and in house from mapping completed by BMT

Figure 4.4 Current benthic habitat coverage within the Local Assessment Unit used at Esperance

4.4.2 Calculating potential losses of benthic communities and habitat

4.4.2.1 Albany

BCH losses were partitioned into three categories: irrecoverable loss (or Zone of High Impact), recoverable loss (Zone of Moderate Impact) and Zone of Influence, representing those habitats exposed to disturbances but with no resulting losses (ZoI). Table 4.8 shows the anticipated areas occupied by each of the three zones. There are no areas of macroalgae within any of the leases.

Table 4.8 Estimated seagrass losses in each Albany Local Assessment Unit

LAU	Current extent of BCH (ha[1])	ZoHI[2] (ha)	ZoMI[3] (ha)	ZoI[4] (ha)
1 - Princess Royal Harbour	1385.00	0.24	13.5	38.19
2 - Inner King George Sound	1159.00	0.09	10.76	55.46
4 - Oyster Harbour	696.44	0.43	60.11	168.71

Notes:
1. ha = hectares
2. ZoHI = Zone of High Impact
3. ZoMI = Zone of Moderate Impact
4. ZoI = Zone of Influence

The area of anticipated permanent loss (ZoHI) was <0.1% of the total area of seagrass mapped within each LAU (Table 4.9). Losses in KGS were estimated at <0.1% taking into account proposed and historical losses. Permanent losses in PRH and OH were 52.07% and 55.07% respectively. Recoverable losses caused by the SCADZ were between 0.47–3.88% of the total pre-European seagrass (Table 4.10). Seagrass habitat in the ZoI was unaffected.

Table 4.9 Estimated permanent seagrass losses in context of historical losses within the Albany Local Assessment Units

LAU	Pre-European mapped area (ha[1])	Historical loss (ha)	Potential permanent loss from SCADZ (ha)	Cumulative area of permanent loss (ha)	% of BCH loss in LAU from SCADZ	% of BCH loss in LAU in total
1 - Princess Royal Harbour	2889.00	1504.00	0.24	1504.24	<0.01	52.07
2 - Inner King George Sound	1159.00	Nil	0.09	0.09	<0.01	<0.01
4 - Oyster Harbour	1549.00	852.56	0.43	852.99	0.03	55.07

Notes:
1. ha = hectares

Table 4.10 Estimated area of seagrasses within zone of moderate impact and zone of influence in context of historical losses within the Albany Local Assessment Units

LAU	Pre-European mapped area (ha[1])	Historical loss (ha)	Area in ZoMI[2] (ha)	% of BCH in ZoMI	Area in ZoI[3] (ha)	% of BCH in ZoI
1 - Princess Royal Harbour	2889.00	1504.00	13.50	0.47	38.19	1.30
2 - Inner King George Sound	1159.00	Nil	10.76	0.93	55.46	4.79
4 - Oyster Harbour	1549.00	852.56	60.11	3.88	168.71	10.89

Notes:
1. ha = hectares
2. ZoMI = Zone of Moderate Impact
3. ZoI = Zone of Influence

4.4.2.2 Esperance

Benthic habitats are not expected to be impacted by abalone ranching, directly or indirectly, as ABITATs are only placed on sandy substrate. As such, no calculations of future losses from the SCADZ were undertaken for Esperance. Taking into account other forms of shellfish farming, irrecoverable losses of BCH will be less than <0.1%.

5 Carrying Capacity

5.1 Overview of studies

5.1.1 Water quality

Water and sediment quality in the Albany region has been the subject of several one-off studies (BMT Oceanica 2014, Hillman 1991, Ecologia 2009 and SKM 2007) and one long term study (DWER 2020) (Table 5.1). The DWER data set comprises some 30 years of data for Oyster Harbour (OH), Princess Royal Harbour (PRH) and KGS and were therefore used in the parameterisation of the model used in this assessment.

Table 5.1 Marine environmental quality studies conducted in Albany region

Study name	Author & year	Location
Albany Port Authority maintenance dredging monitoring	BMT Oceanica (2014)	Princess Royal Harbour and King George Sound
Albany Iron Ore Project Public Environmental Review – Albany Port Expansion Proposal EPA Assessment No 1594	Ecologia (2009)	Princess Royal Harbour and King George Sound
Sampling and Analysis Plan Report for the Albany Iron Ore Project	SKM (2007)	Princess Royal Harbour and King George Sound
DWER water monitoring	DWER (2020)	Oyster Harbour, Princess Royal Harbour and King George Sound

5.1.2 Historical application of models

Several studies have examined the carrying capacity of shellfish aquaculture e.g. Congleton et al. (1999), Arnold et al. (2000), Chamberlain (2006), Crawford et al. (1996); Joyce et al. (2010); Ferreira et al. (2008). Previous applications varied from simple physical and/or production models (Congleton 1999, Arnold 2000, Chamberlain 2006) through to fully integrated ecological models (Ferreira et al. 2008) (see definitions in McKindsey et al. 2006) (summarised in Table 5.2).

Table 5.2 Types of carrying capacity models applied to shellfish farms

Model	Description
Physical	Based on the total area of marine farms that can be accommodated in the available physical space. Typically assessed using a combination of hydrodynamic models and physical information, and ideally presented and analysed within a Geographic Information System (GIS).
Production	Based on the stocking density of bivalves at which harvests are maximised. Typically assessed with consideration of the available food resources, which ultimately relates to the productivity and the functioning of the ecosystem.
Ecological	Based on the stocking or farm density which causes unacceptable ecological impacts. Ecological models take a more holistic approach by considering the broader ecological requirements of the water body.
Social	Based on the level of farm development that causes unacceptable social impacts. The social carrying capacity is even more complex than the ecological carrying capacity. It comprises the above three categories (physical, production and ecological) as well as the trade-offs between all stakeholders to meet the demands of the population (socioeconomic factors such as traditional fisheries, employment and recreational use) and the environment.

Source: McKindsey et al. (2006)

Physical models involve the application of Geographic Information Systems (GIS) and attempt to determine the siting of farming zones relative to competing interests such as recreational and or industrial areas (particularly proximity to wastewater disposal) e.g. Arnold et al. (2000). Siting is typically based on a weighted statistics approach which determines the relative suitability of proposed farming zones based on criteria such as water quality, current speed, wave climate, recreational needs and environmental criteria (e.g. Congleton et al. 1999). While suitable for planning purposes, physical models do not consider nutrient availability, feeding requirements nor the potential for broader ecological impacts to the system.

Production models depend on the physical carrying capacity and functions of the ecosystems, especially primary production. However, production models are limited in that they fail to consider the feedback mechanisms between the culture activities and the needs of the ecosystem. For this reason, ecological models have proven popular in western cultures, where there are established environmental regulatory frameworks, competing socio-environmental interest and strong conservation principles e.g. Crawford et al. (1996); Joyce et al. (2010); Ferreira et al. (2008).

Although applied in multiple jurisdictions, there is also a perception that ecological models are overly simplistic, because they fail to account for the many nuances that ultimately affect carrying capacity. The NSW Oyster Industry Sustainable Aquaculture Strategy for example notes that there is currently insufficient data to accurately quantify optimal stocking densities from a carrying capacity perspective (Government of NSW 2021). Difficulties arise when developing ecological carrying capacity models due to the inherent variability in ecosystems, where in-situ information is not available in sufficient detail. As such, the interpretability of the results can be limited.

However, ecological carrying capacity models have also been applied successfully, both with and without extensive validation. Grant et al. (2005) and Ferreira et al. (2008) for example demonstrated broad agreement between their models and in situ measurements of phytoplankton drawdown and shellfish bio-energetics. While the Ferreira et al. (2008) model benefited from extensive validation, the model applied by Grant et al. (2005) relied on the intelligent application of published mussel clearance and feeding rates, together with a hydrodynamic model to simulate system flushing.

5.2 Potential impacts

The potential for direct and indirect impacts due to exceeding the ecological carrying capacity of a system are summarised in Table 5.3Table 6.1. Indirect impacts are system specific and difficult to define due to complex trophic cascades. Direct impacts are limited to declines in suspended organic matter and phytoplankton. Other potential direct impacts may include changes to phytoplankton assemblages due to preferential feeding habits of some shellfish. However, like the indirect effects described above, the multivariate nature of such changes, combined with significant levels of natural variability, render such changes difficult to detect. For this reason, the carrying capacities of OH, KGS and PRH were conservatively assessed based on projected declines in phytoplankton biomass (see Section 5.3.7).

Table 5.3 Potential direct and indirect impacts of exceeding the carrying capacity

Potential impacts	Context
Direct impacts	• Declines in phytoplankton biomass to levels below the seasonal baseline • Declines in organic matter to levels below the seasonal baseline
Indirect impacts	• Declines in both farmed and natural occurring shellfish health Competition with other filter feeders, increasing recycling speed of nutrients, removal of eggs and larvae of fish and benthic organisms • Subsequent changes in the ecological function due to trophic cascades

5.3 Methods

5.3.1 The SCADZ model

For this study, BMT employed AquaDEEP, a one-dimensional hydrodynamic model coupled to a simple ecological model, building on the approaches of Crawford et al. (1996) and Joyce et al. (2010). Additions to our model included the simulation of calendar seasons, and the inclusion of some bio-energetic function with three clearance rate options (minimum, medium and maximum). Although including simple bio-energetic functionality (i.e. clearance rate variation), our model lacked the experimental data needed to validate the bio-energetics component of the model (as applied successfully by Ferreira et al. 2008).

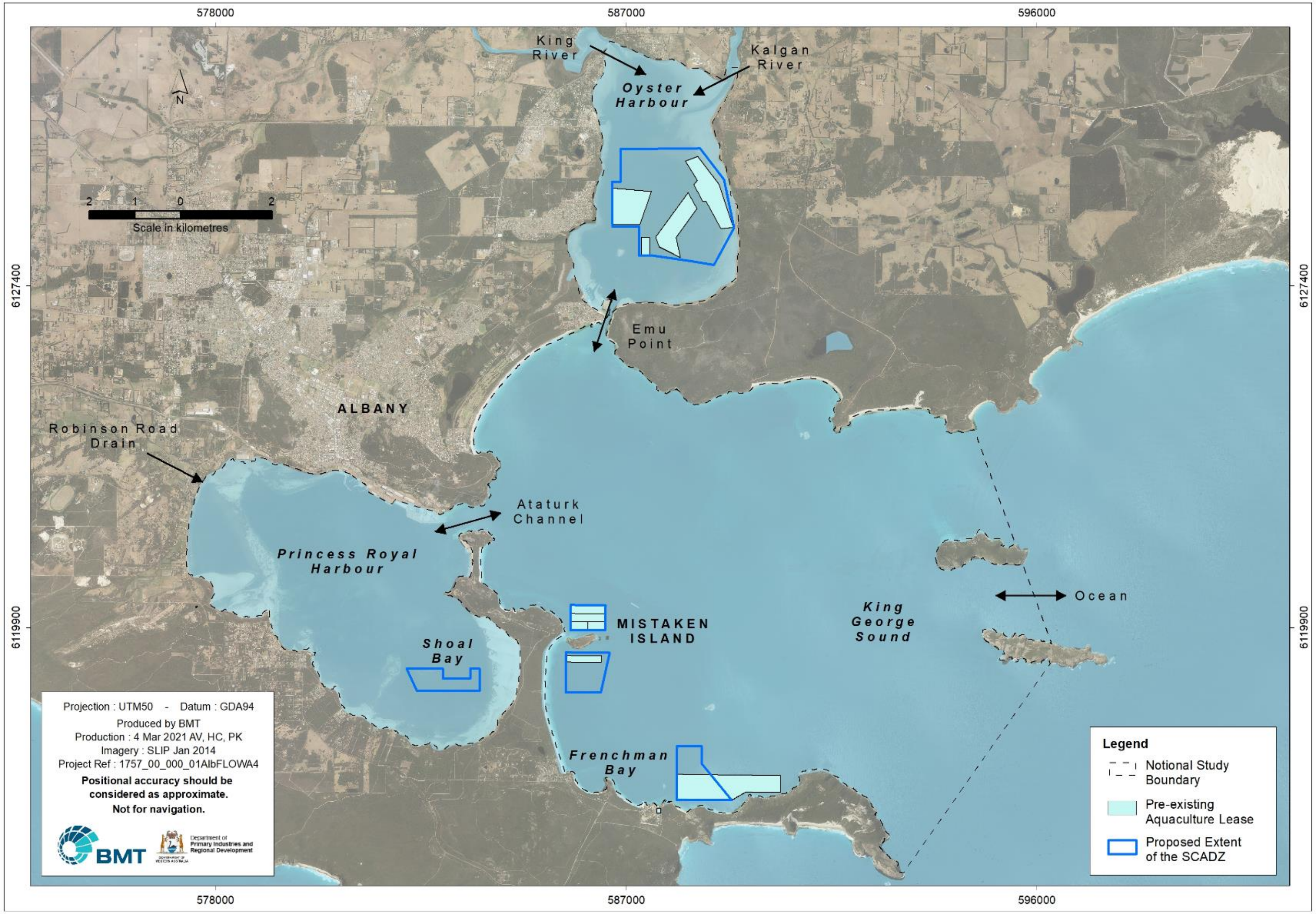
AquaDEEP was used to conservatively estimate the carrying capacity of the OH, KGS and PRH based on multiple production scenarios (see Section 5.3.8). Carrying capacity was defined as the standing biomass that could be sustained, without reducing regional phytoplankton biomasses to levels below 1 µg.chlorophyll-a/L, which is the threshold for healthy oligotrophic ecosystems in southwestern Western Australia (Brearley 2005). Final carrying capacities are provided as upper and lower estimates to account for the variably and uncertainty in the calibration data.

5.3.2 Model architecture

To estimate the carrying capacity of the SCADZ, the region was partitioned into three areas (Figure 5.1): OH; PRH and KGS. For each area, a coupled hydrodynamic (GLM) and water quality (AED2) model was applied to define:

- The existing nutrient budget (CNB) - based on the contemporary shellfish and biofouling biomass and
- The sustainable nutrient budget (SNB) - based on a sustainable theoretical carrying capacity within OH, PRH and KGS as defined in the model.

The linkage between the hydrodynamics and water quality is important as the hydrodynamic model simulates water balances and vertical water density stratification, which in turn influences the biochemical processes simulated by the water quality model. The carrying capacity is the standing biomass present at a given time, as simulated by the model. Further details regarding the modelling architecture (GLM-AED2) are provided in Appendix B.



Notes:
1. Single arrows indicate the main inputs of nutrients and water into each area
2. Double arrows indicate the exchange of nutrients and water in each area
3. The dashed line indicates the notional boundary of the study area

Figure 5.1 The modelled areas, encompassing Oyster Harbour, Princess Royal Harbour and King George Sound

5.3.3 Boundary Conditions

Boundary conditions (BC) are the data used to force the model. The BCs for this study were derived from long-term catchment inflows and loads, ocean exchanges and meteorological data (Table 5.4). The King and Kalgan Rivers contribute significant freshwater flows and nutrient loads to OH, particularly in the winter months. Inflows for both rivers were obtained using DWER's Water Information Reporting Database (DWER 2020) from stations 602014 and 602004, respectively. The main inflows and loads to PRH are the Robinson Road and Munster Hill drains (WIR Station 602010). Data obtained from BMT (2014) were used to fill gaps in the inflow estimates.

Table 5.4 Summary of the data used to force the model

Location	Data Type	Source
Entire model domain	Meteorological	BoM Station 009999 (Albany Airport)
King River	Flows	WIR[1] Station 602014 and 602015
King River	WQI[2]	WIR Station 602015
Kalgan River	Flows	WIR Station 602004
Kalgan River	WQI	WIR Station 602004
Robinson Road and Munster Hill drains	Flows	BMT 2014 WIR Stations 602009 and 60210
Robinson Road and Munster Hill drains	WQI	WIR Station 602010 Supplemented by BMT 2014
Ocean flow exchange at Emu Point, between Princess Royal Harbour and King George Sound and between King George Sound and open ocean	Daily inflows and outflows	Calculations based on data obtained by BMT in 2008 and global circulation model
Emu Point	WQI	WIR Station 6021174 (most complete station close to Emu Point)
Ocean input to King George Sound	WQI	Based on global ocean review papers, as all stations are located very close to the shore
Ataturk Channel	WQI	WIR Station 6021172 (most complete station close to Ataturk channel)
The entire area of interest	Bathymetry	Navionics BMT data collected previously

Notes:
1. WIR = Water Information Reporting database
2. WQI = Water Quality Indicators. Note that where WQI could not be found at the WIR stations or at site specific literature, generic literature values were used (e.g. TOC at the ocean boundary condition)

5.3.4 Calibration

To establish a baseline comparative index, model parameters were adjusted until model predictions matched the observed range and the seasonal and interannual water quality variability in each of the study areas. The data used in the calibration comprised nutrient and phytoplankton (chlorophyll-a) concentrations as well as temperature and salinity values, in combination with raw and published data within each of the three areas of interest. The values were calibrated across an estimated period of the last seven years to capture seasonal and inter-annual variability. This specific time period was chosen as it had the most recent and consistent data available.

The range of values recorded by DWER and historically by Hillman (1991) are presented in Table 5.5. The values highlighted in grey were deemed most representative and were used as aspirational targets for the calibration. The performance of the model relative to the calibration is represented in Figure 5.2 to Figure 5.4. The pink broken lines represent the natural minimum and maximum values recorded. In the same figures, the blue line represents the model predictions.

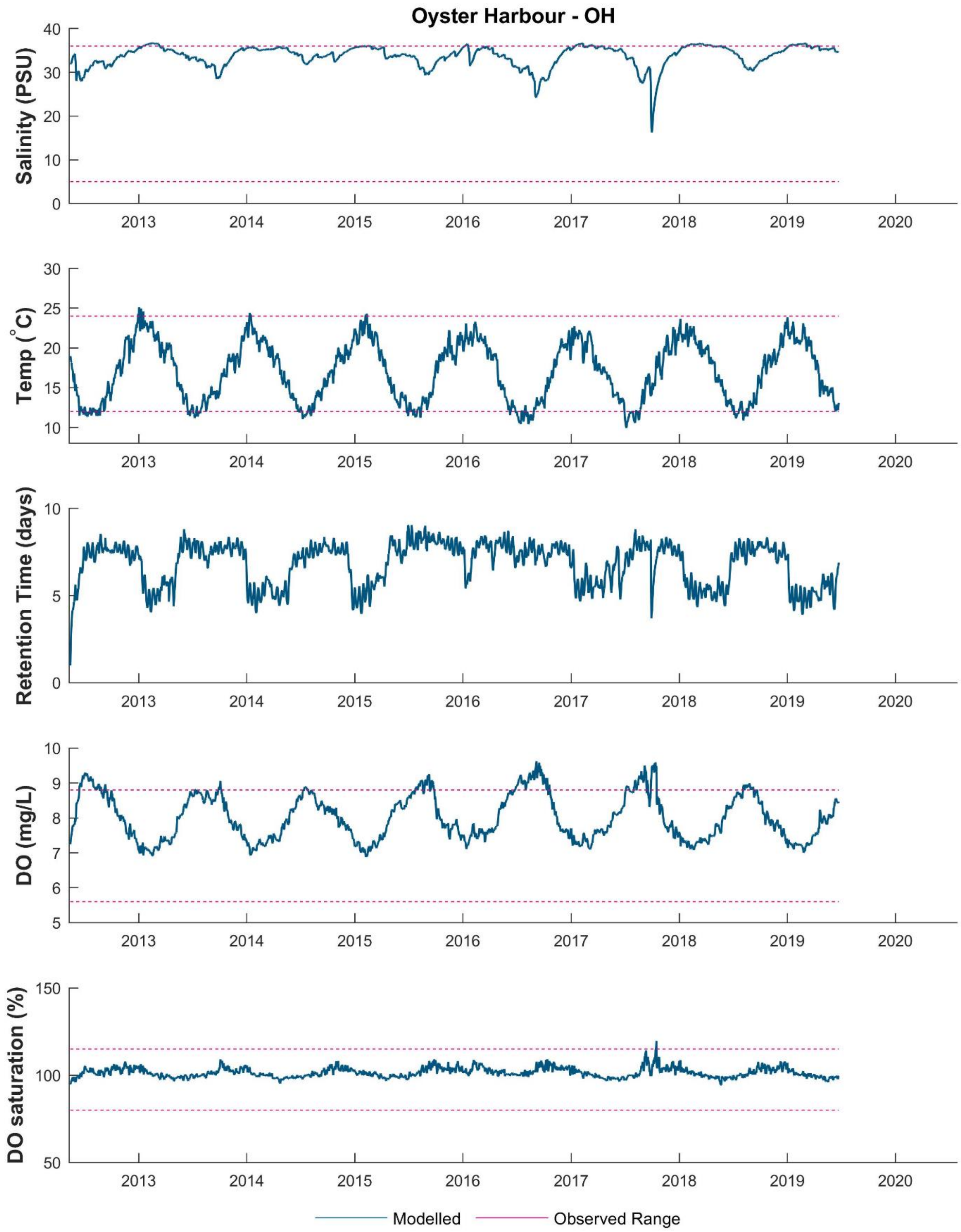
These predictions fall generally within the natural range, indicating a good degree of fit and therefore adequate model calibration for the baseline index.

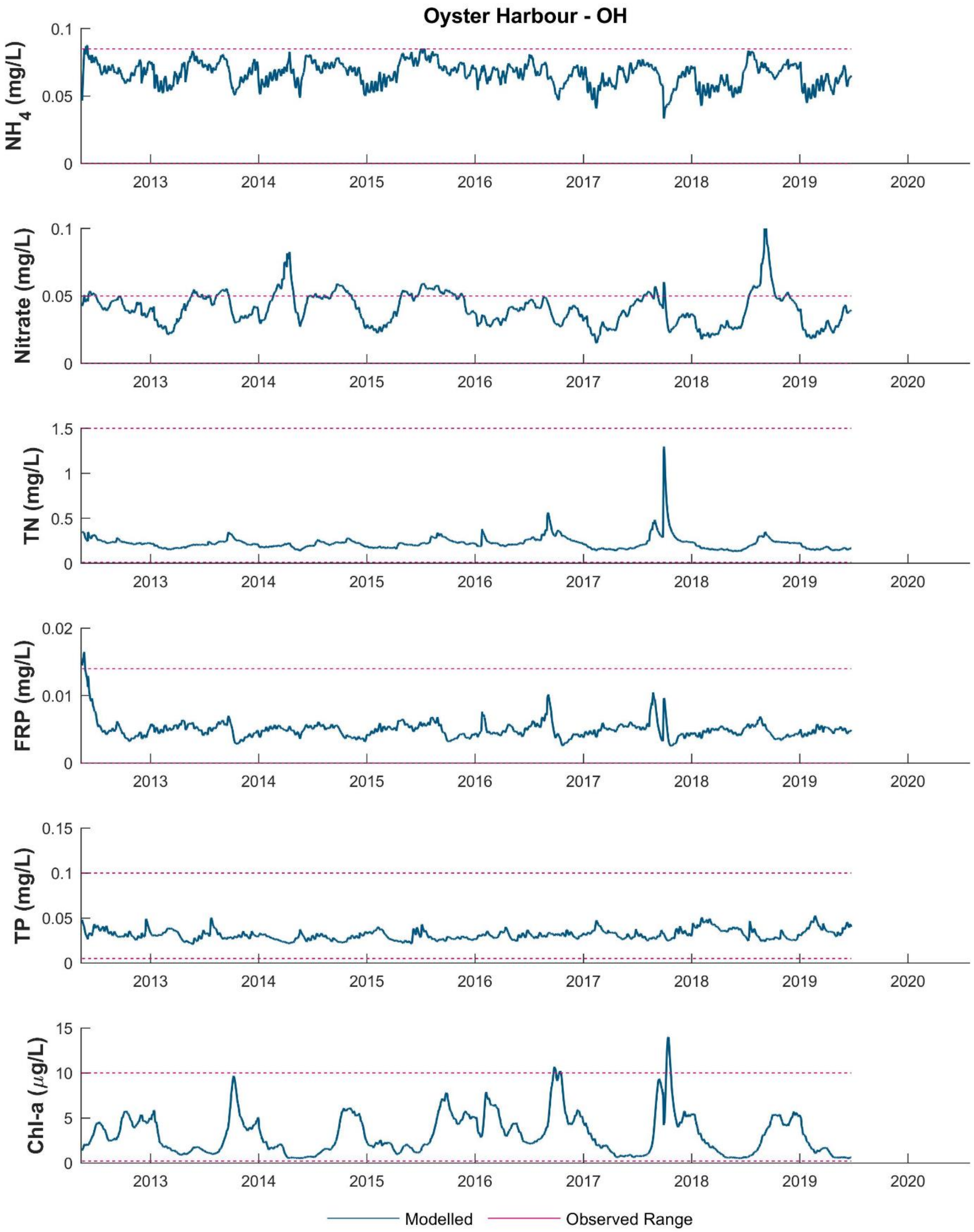
Table 5.5 Summary of the values used to guide model calibration

Variable	Hillman (1991)	WIR Data (1988–2019 / intermittently)	DWER (2016–2017) (Thomson, 2018)	WIR Stations
Temperature	17–22[1] °C		12–24 °C	Oyster Harbour: 6021173 6021175 6021176 6021177 6021257 6021258 6021259 6021260 6021291 6021436 6021437 6021438 6021446 6021447
Salinity	25–37 psu		5–36 ppt	
DO	5.6–8.8 mg/L / 70–110%[2]	0.04–16.05 mg/L 0.5–200% saturation	3.4–12 mg/L	
Phosphate	0–0.080 mg/L		0.002–0.016 mg/L	
Ammonium	0–0.140 mg/L		0.002–0.085 mg/L	
Nitrate	0–0.300 mg/L		0.0001–0.05 mg/L	
TON	0.010–0.800mg/L		0.1–1.3 mg/L	
TN	0.01–1.24 mg/L	0.05–3.1 mg/L	0.01–1.5 mg/L	
TOP	0.025–0.080 mg/L			
TP	0.025–0.18 mg/L	0.005–0.87 mg/L	0.005–0.87 mg/L	
Silicate	0.250–1.700 mg/L			
Chl-a	0.5–3.5 µg/L	0.2–11 µg/L	0.2–11 µg/L	
Temperature	14–22 °C			Princess Royal Harbour: 6021168 6021169 6021170 6021453 6021454 6021455 6021456 6021463 6021464 6021678 6021685 6021171
Salinity	32–37 psu			
DO	6.4–8.8 mg/L 80–110% saturation	4.5–13 mg/L 59–161.4% saturation		
Phosphate	0.0–0.012 mg/L			
Ammonium	0.005–0.050 mg/L			
Nitrate	0–0.010 mg/L	0–0.05 mg/L		
TON	0.020–0.220 mg/L			
TN	0.025–0.25 mg/L	0.025–0.5 mg/L		
TOP	0.010–0.030 mg/L			
TP	0.01–0.04 mg/L	0.0050–0.0600 mg/L		
Silicate	0.010–1.500 mg/L			
Chl-a	0. 5–1 µg/L	0.3–3 µg/L		
Temperature	15–24 °C			King George Sound: 6021444 6021178 6021275 6021179
Salinity	33–36 psu			
DO	6.5–9.2 mg/L 80–115% saturation	7.2–8.8 mg/L 96–117 % saturation		

Variable	Hillman (1991)	WIR Data (1988–2019 / intermittently)	DWER (2016–2017) (Thomson, 2018)	WIR Stations
Phosphate	0–0.005 mg/L			6021677
Ammonium	0–0.020 mg/L			
Nitrate	0–0.010 mg/L	0.005–0.018 mg/L		
TN	0.005–0.200 mg/L	0.02–0.12 mg/L		
TP	0.010–0.055 mg/L			
Silicate	0.010–0.050 mg/L			
Chl-a	0–1 µg/L	0.05–0.5 µg/L		

Notes:
1. Note that the range refers to the minimum and maximum ever reported / observed
2. The values highlighted in grey represent the targeted value for calibration

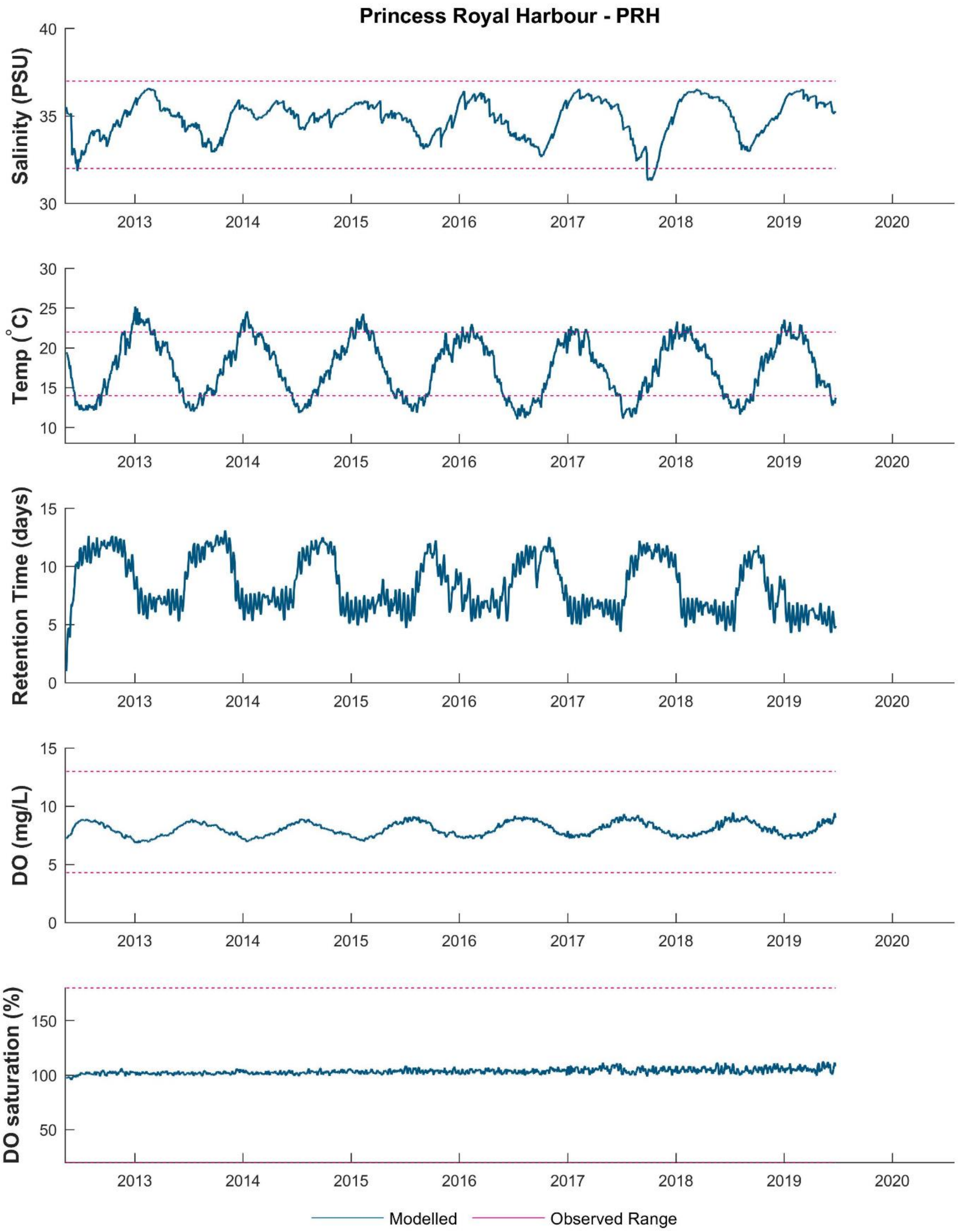

Oyster Harbour - OH
Salinity (PSU)
Temp (°C)
Retention Time (days)
DO (mg/L)
DO saturation (%)
2013
2014
2015
2016
2017
2018
2019
2020
Modelled
Observed Range

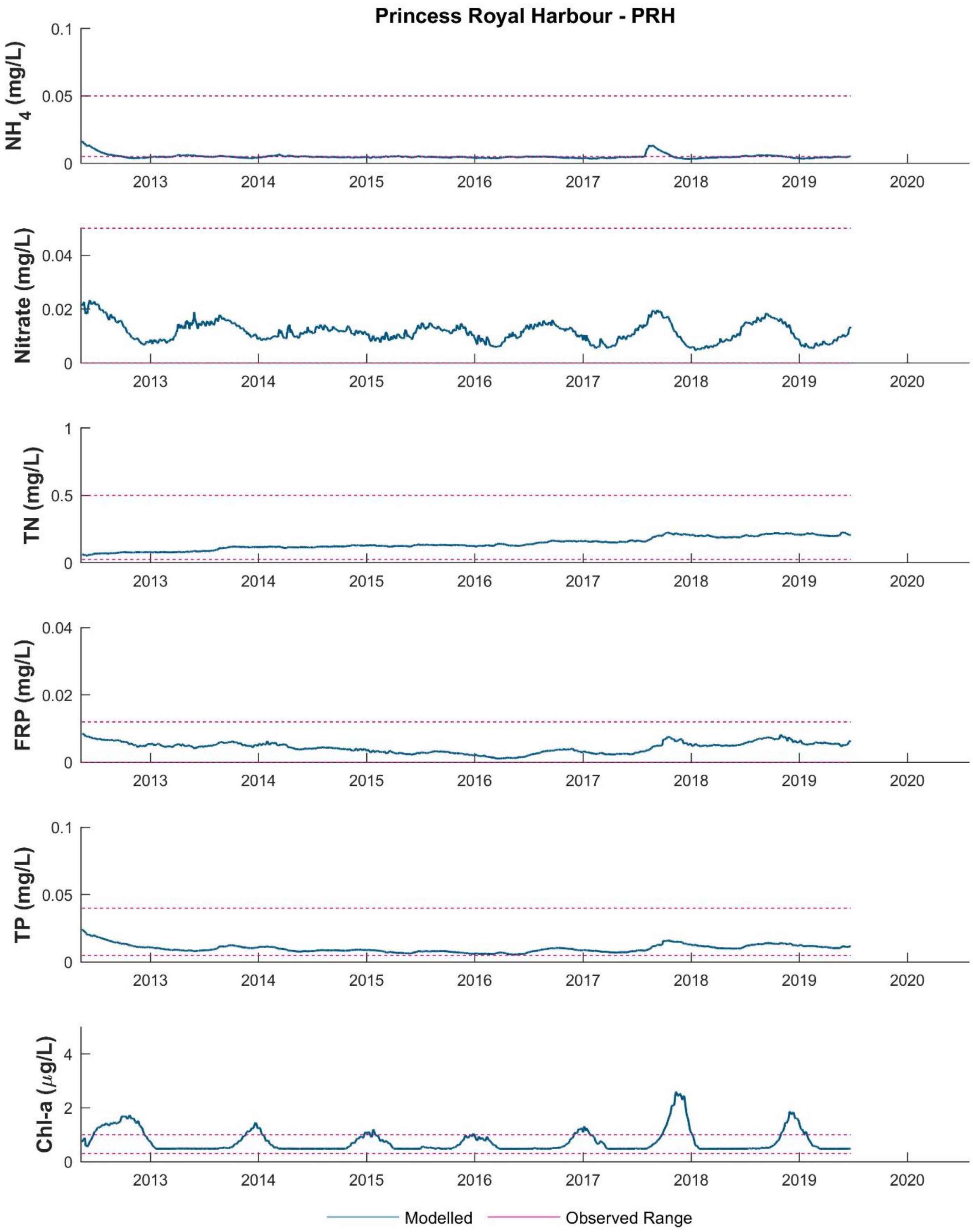


Notes:

1. Pink dashed lines represent the maximum and minimum value on record. Surface refers to the first 2 m depth

Figure 5.2 Water quality indicators Oyster Harbour between 2012-2019

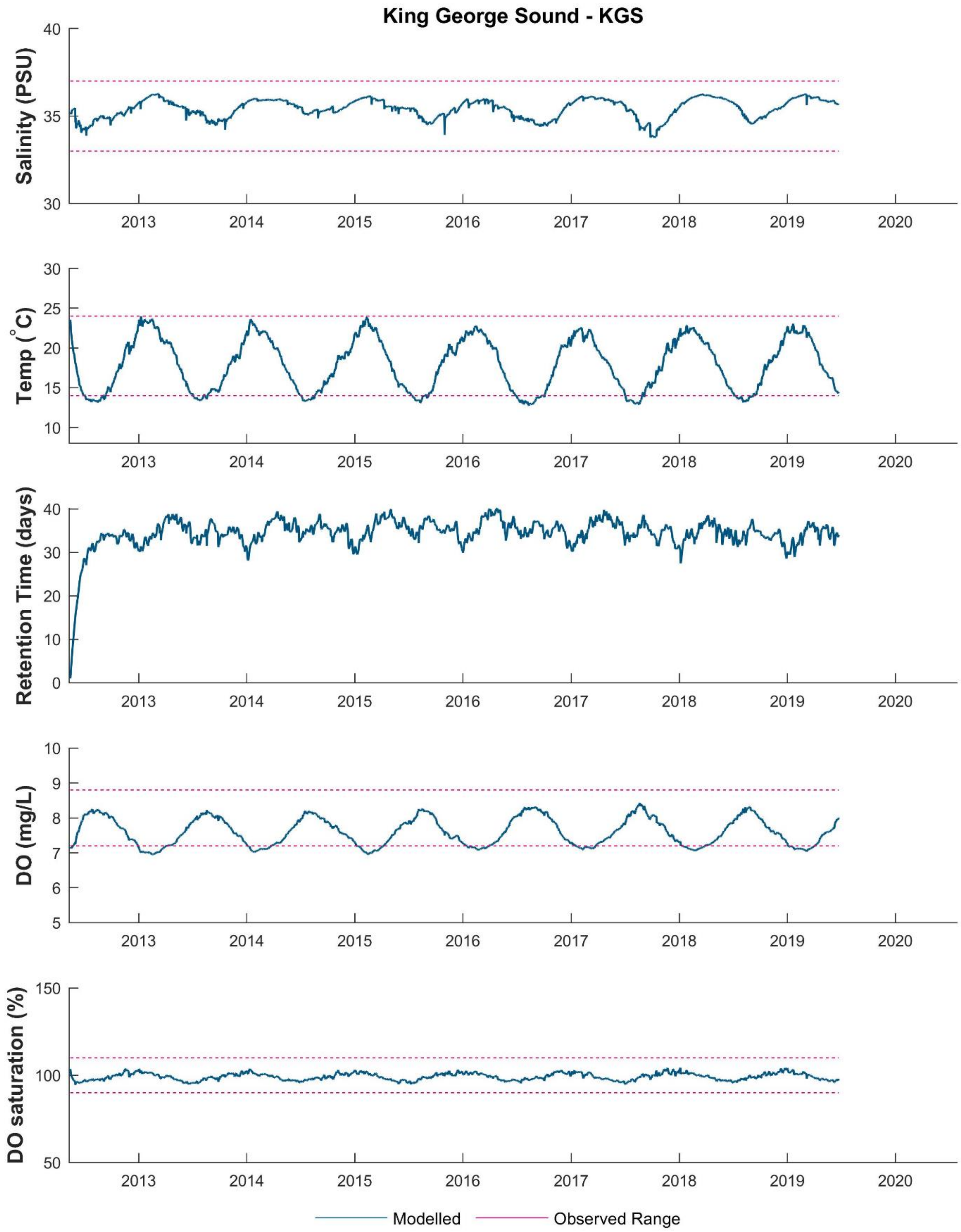

Princess Royal Harbour - PRH
Salinity (PSU)
Temp (°C)
Retention Time (days)
DO (mg/L)
DO saturation (%)
2013
2014
2015
2016
2017
2018
2019
2020
Modelled
Observed Range

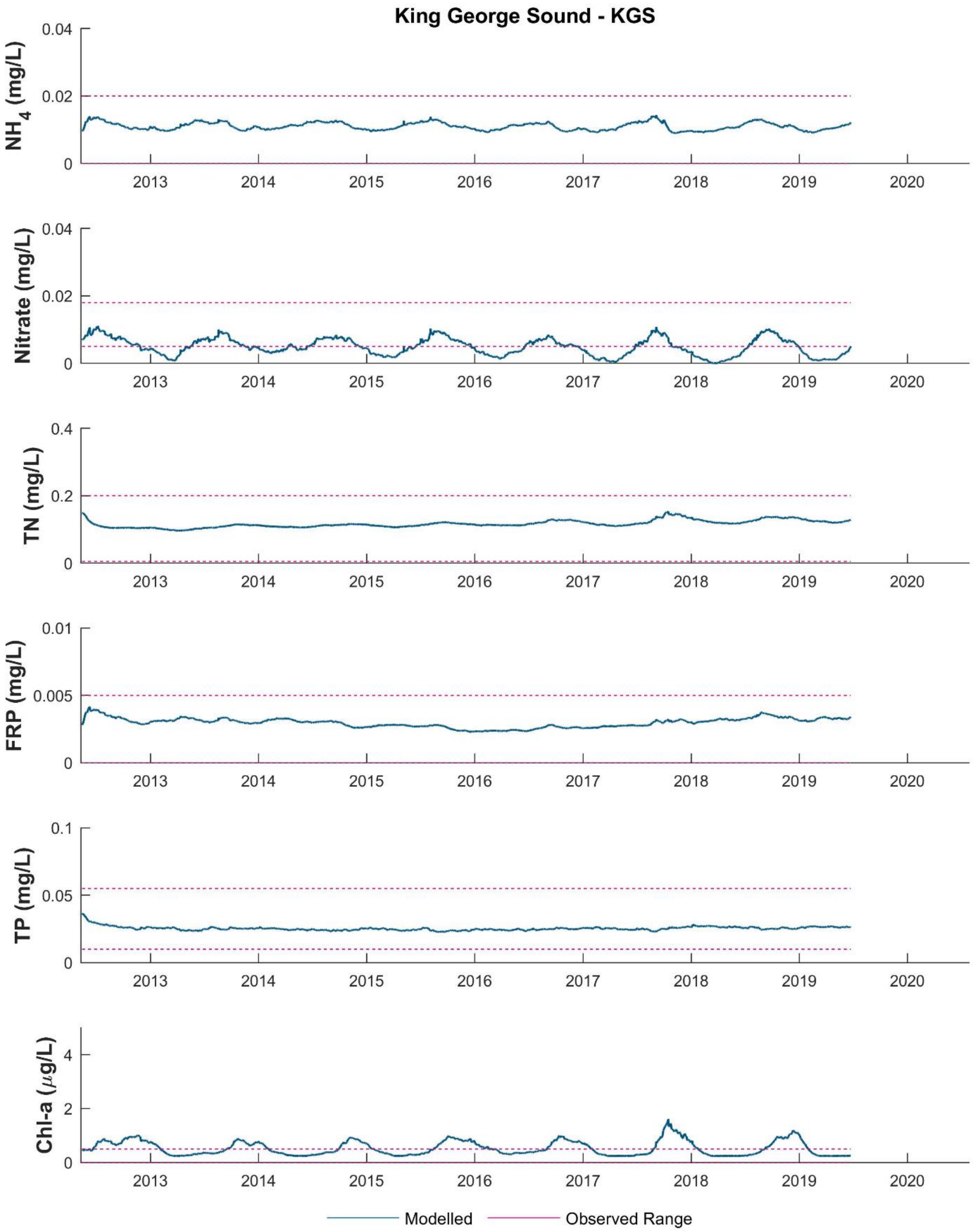


Notes:

1. Pink dashed lines represent the maximum and minimum value on record. Surface refers to the first 2 m depth

Figure 5.3 Water quality indicators for Princess Royal Harbour between 2012-2019


King George Sound - KGS
Salinity (PSU)
Temp (°C)
Retention Time (days)
DO (mg/L)
DO saturation (%)
2013
2014
2015
2016
2017
2018
2019
2020
Modelled
Observed Range



Notes:
1. Pink dashed lines represent the maximum and minimum value on record. Surface refers to the first 2 m depth

Figure 5.4 Water quality indicators for King George Sound between 2012-2019

5.3.5 Baseline shellfish biomasses

Before running the carrying capacity scenarios, it was first necessary to establish the environmental baseline conditions for each of the OH, KGS and PRH systems. Baseline conditions were modelled based on the current biomasses of farmed shellfish, plus the estimated contribution of biofouling attached to aquaculture infrastructure. Biofouling is the naturally occurring biological materials – such as mussels, algae, barnacles, ascidians, sponges, worms and sea cucumber – that attaches itself to the aquaculture infrastructure (Figure 5.5).

Present day biomasses were set based on production and sales records for the period 2016-2017. The present-day biomass of *S. glomerata* in OH was calculated based on seeding records provided by the Albany shellfish industry. Current adult biomasses used in the model were extrapolated based on the number of juveniles supplied to OH, and their projected growth rate and mortality over a three-year period. Baseline biomasses for *M. galloprovincialis* and *P. fucata* at Mistaken Island were derived from official production records provided by industry, as were the baseline biomasses for *M. galloprovincialis* in OH.

Baseline shellfish biomasses for PRH and FB were set at zero for this assessment, though it was noted that both have carried significant biomasses in the past, especially at FB (Table 5.6). Further anecdotes collected during our site visit (June 2021) indicated that all of the modelled areas previously carried substantial shellfish (and biofouling) biomasses, particularly during the 1990's. While not used to set the baseline biomasses, these anecdotes were noted as relevant historical records (Table 5.6).

5.3.6 Biofouling contribution

Farming infrastructure including posts, ropes, floats and oyster baskets provide suitable settlement points and refuges for an array of biofouling organisms. Evidence for the extent of present-day biofouling in OH was collected on a site visit in June 2021 (Table 5.6, Figure 5.5). Observations were that biofouling was conservatively 50 to 70% of the farmed shellfish biomass, with the ability to settle and grow rapidly immediately following the deployment of baskets or longlines (Figure 5.5). Biofouling organisms comprised two acorn barnacle species (Family Balanidae), holothurians, ascidians, native pearl oyster *Electroma papilionacea* as well as *M. galloprovincialis* when present on *S. glomerata* or *P. fucata* infrastructure. Barnacles are the predominant biofouling organism in OH, while the other filter feeders, primarily *E. papilionacea*, are more common at Mistaken Island.

The contribution of biofouling to the system was modelled under the assumption that it represented 50% of the biomass of farmed shellfish under baseline conditions, and 50% of future production scenarios i.e. for every 1 tonne increase in biomass of farmed shellfish, we added 0.5 tonnes of biofouling organisms. Biofouling biomasses were modelled to remove particles from the water column (i.e. feeding rate) at a rate of 1 L g^{-1} h^{-1}, or roughly half that of the medium clearance rate for *S. glomerata* and *M. galloprovincialis,* based on Kohan et al. (2019).

An important assumption in the model was that none of the biofouling was removed from the system, but returned to the water during routine cleaning. Under this assumption, all of the carbon, nitrogen and phosphorus was remobilised and made available for new phytoplankton growth. By contrast all of the nutrients accumulated by oysters and mussels were removed from the system upon harvesting.

Table 5.6 Anecdotal evidence of biofouling and historical shellfish biomasses

Anecdote source	Subject	Anecdote(s)
Robert Michael: Harvest Road	Biofouling	• In OH, oyster baskets typically house a thick under-side layer of barnacles (Figure 5.5). • Barnacle growth is strongest in summer and weakest in winter. • Summer growth rates are prolific. Figure 5.5 shows approximately two weeks of growth post basket deployment. • In the 1990s, it was not uncommon for mussel lines to contain 5-7kg of product per metre plus another 10kg of biofouling.
Jonathan Bilton: Albany Shellfish Hatchery. Previously shellfish farmer in Oyster Harbour and King George Sound	Historical shellfish biomasses	• *M. galloprovincialis* were farmed in FB between 1992 and 2017. • During that time, standing biomasses of shellfish reached a maximum of 1000 tonnes. • Farmed shellfish standing biomasses in OH reached a maximum of potentially 1000 tonnes in the 1990s. • Shellfish were previously farmed in PRH, but were never farmed extensively.
	Biofouling	• Historical aquaculture operations in OH (1990s) used a combination of catching (for catching natural spat in the system) and grow-out rope (for growing out the caught spat to marketable size). • Anecdotally, farmers deployed 100 kms of catching rope, of which only 30 kms was viable. The remaining 70 kms became so extensively bio-fouled that the weight of the shellfish biomass caused it to slough off before the farmers could retrieve it.
Gillies et al. 2018 – Australian shellfish ecosystems: Past distribution, current status and future direction	Historical shellfish biomasses	• OH once housed extensive reefs of the native flat oyster *Ostrea angasi*, however the extent or total biomass have never been quantified.

Source: BMT

Figure 5.5 Indicative extent of barnacle biofouling on oyster baskets two weeks post-deployment in Oyster Harbour

5.3.7 Impact thresholds

For this study, BMT employed AquaDEEP, a one-dimensional hydrodynamic and water quality model, building on the approaches of Crawford et al. (1996) and Joyce et al. (2010). AquaDEEP was used to conservatively estimate the carrying capacity of the OH, KGS and PRH based on multiple production scenarios. Carrying capacity was defined as the standing biomass that could be sustained, without reducing regional phytoplankton biomasses to levels below 1 µg.chlorophyll-a/L, which is the threshold for healthy oligotrophic ecosystems in southwestern Western Australia (Brearley 2005).

5.3.8 Scenarios

Carrying capacity estimates for OH, KGS and PRH were derived from multiple shellfish production scenarios (i.e. shellfish tonnages produced annually). In total, six scenarios were modelled for OH, three for KGS and three for PRH. OH was examined in greater detail because it is likely to accommodate greater shellfish production. To accommodate the uncertainty in feeding rates, published minimum, medium and maximum clearance rates were modelled for each shellfish biomass scenario (Table 5.7).

When established, it is likely that PRH will be used as a depuration site to supplement OH during seasonal closures, rather than as a grow-out area. In this sense, shellfish from OH may be moved to PRH for six months of the year especially during WASQAP enforced closer periods. Moving stock between the two sites will ensure aquaculture operators can still provide shellfish to market all year round. The use of PRH in this manner means that its productivity (i.e. annual production of shellfish) is not a focus for aquaculture operators. For this study however, PRH was modelled in the context of annual production to maintain consistency with the other areas modelled, and to provide a starting point in the event it is ever used for shellfish grow-out in the future.

The modelled scenarios were based on production rates supplied by DPIRD, industry (Table 5.7) and the Australian literature. The time taken to reach harvestable maturity was estimated at three years for *S. glomerata*, and one year each for *M. galloprovincialis* and *P. fucata*. Mean values for clearance rate are summarised in Table 5.7.

Mean bio-deposition rates as derived from the literature were 10% for *S. glomerata* and *P. fucata*, and 7.5% for *M. galloprovincialis* of the dry tissue weight. Mean ammonia excretion rates as derived from the literature were 7.2% of the dry tissue weight for each species.

Modelling utilised an iterative process whereby the initial GLM results were run for each area several times to simulate flow exchange between each of the three areas. Additionally, further modelling of three scenarios from OH, and all scenarios for PRH and KGS, was conducted assuming zero production levels in the other two areas (i.e. with no depletion). This was undertaken to illustrate how increasing biomass in one area affects food availability in others. The process aims to highlight the importance of the SCADZ as an inter-connected system, rather than assessing each area individually.

Table 5.7 The scenarios and associated parameters run to estimate the carrying capacity of shellfish in each of the Albany assessment areas

Location	Shellfish species	S1			S2			S3			S4			S5			S6			Clearance rate (L g^{-1} h^{-1})[4]	References
		SD[1]	SB[2]	AP[3]	SD[1]	SB[2]	AP[3]	SD[1]	SB[2]	AP[3]	SD[1]	SB[2]	AP[3]	SD[1]	SB[2]	AP[3]	SD[1]	SB[2]	AP[3]		
Oyster Harbour	Sydney rock oyster (*Saccostrea glomerata*)	0.56	302	100	0.83	448	150	1	594	198	1.67	902	300	5	2700	900	8.34	4500	1500	1.3	Bayne et al. 1999; Cranford et al. 2011
																				2.2	
																				4	
Princess Royal Harbour[5]	Sydney rock oyster (*S. glomerata*)	0.67	40	14	1	60	20	1.33	80	27										1.3	
																				2.2	
																				4	
King George Sound	Akoya pearl oyster (*Pinctada imbricata fucata*)	0.76	68	68	1.15	104	104	1.54	139	139	Not applicable									5	Cranford et al. 2011; Numaguchi 1994
																				15	
																				30	
	Blue mussel (*Mytilus galloprovincialis*)	2	236	236	3	354	354	4	472	472										1.3	Cranford et al. 2011; Pascoe et al. 2009
																				2	
																				4	

Notes:
1. SD = Stocking density representing the overall tonnage per hectare of shellfish for each area. Assumes the aquaculture infrastructure is equally distributed across the entire area available for leases (e.g. for OH across the entire 540 ha). Actual stocking densities may fluctuate beyond these numbers within final approved leases.
2. SB = Standing biomass representing the total biomass within a lease, given in tonnes, calculated by multiplying the stocking density by the total ha of a lease. These numbers do not include current farmed biomasses.
3. AP = Annual production estimating tonnes of shellfish produced annually in each area. For *S. glomerata*, annual production was estimated to be a third of standing biomass, as it takes approximately three years for spat to grow to harvestable adults. For *P. fucata* and *M. galloprovincialis*, annual production was estimated to be the same as standing biomass as it takes approximately one year for spat to grow to harvestable adults. These numbers do not include current farm productions.
4. Clearance rates are standardized per 1 g of dry tissue weight. The three different values represent minimum, medium and maximum clearance rates for each shellfish species as determined from a literature review.
5. Though Princess Royal Harbour will likely be used as a depuration rather than a grow-out site (see Section 5.3.8), for the purpose of this study annual production was still considered to maintain consistency with other modelled areas.

5.3.9 Model Assumptions

The assumptions underpinning the carrying capacity assessment are provided in Table 5.8.

Table 5.8 Assumptions underpinning modelling of carrying capacity

Assumption
• Each area (OH, PRH and KGS) was represented individually, and an iterative process was used to generate the exchange of salts and nutrients between them (i.e. they were not dynamically linked). • Areas were modelled individually and concurrently to verify the extent of connectivity between the areas.
• All three areas were simplified into a one-dimensional systems i.e. no horizontal variations were resolved.
• Ocean inflow temperatures are based on 2018, when BMT collected data in King George Sound.
• When individual nutrient measurements from the catchments and ocean were missing in the long-term timeseries, the median of the available data was used to generate the input to the model.
• Boundary conditions for chlorophyll-a for the oceanic boundary in relation to inflows to PRH were changed from 0.125 to 0.5 μg chlorophyll-a/L based on newly available knowledge. As such, modelled chlorophyll-a levels for PRH could not be reduced to below 0.5 μg chlorophyll-a/L.
• Clearance, bio-deposition and excretion rates are based on the literature. Further refinement maybe achieved by replacing these data with in-situ measured data.
• Shellfish were simulated as a static sink and source of nutrients i.e. their bioenergetics were not simulated dynamically in response to ambient environmental conditions. The exception to this was a temperature limitation at 12°C, whereby shellfish metabolism declined to zero below this temperature.
• Shellfish were simulated feeding and egesting at the rate of adult sized (~50 g for *S. glomerata* and *P. fucata*, and ~40 g for *M. galloprovincialis*). All standing biomass tonnages presented in the report assume adult specimens. • Conversion of standing biomass to production tonnages for *S. glomerata* was achieved by dividing the standing biomass by three, based on an assumed three-year grow out period. This ignores the growth of the animals over time and assumes oysters were deployed at adult size and weight (for calculation of clearing efficiencies).
• Clearance efficiencies were simulated at 80% efficiency for all species, based on Cranford et al. (2011) and Pascoe et al. (2009).
• The percentage submersion time for *S. glomerata* was set at 75% to match the estimated exposure time over their three-year grow-out cycle.
• Baseline shellfish biomasses were estimated based on industry records from 2017. See Section 5.3.5.
• The extent of biofouling included in the models was determined based on in-situ evidence provided by shellfish operators (photos, interviews, historical references) during a site visit. • Biofouling was modelled under the assumption that it represented 50% of the biomass of farmed shellfish under baseline conditions, and 50% of future production scenarios i.e. for every 1 tonne increase in biomass of farmed shellfish, we added 0.5 tonnes of biofouling organisms. • Biofouling biomasses were modelled to feed at a rate of 1 L g^{-1} h^{-1}, or roughly half that of the medium clearance rate for S. glomerata and M. galloprovincialis, based on Kohan et al. (2019). • An important assumption in the model was that none of the biofouling was removed from the system, but returned to the water during routine cleaning. Under this assumption, all of the carbon, nitrogen and phosphorus was remobilised and made available for new phytoplankton growth. By contrast all of the nutrients accumulated by oysters and mussels were removed from the system upon harvesting.
• Barnacles are the dominant form of biofouling in OH and *E. papilionacea* in MI. Though barnacles feed on a range of food sources (e.g. zooplankton), phytoplankton is still their primary food source. Although other biofouling organisms (e.g. ascidians) were considered, they were found not in direct competition with farmed shellfish, given their preferences for other food sources (Kang et al. 2009).
• Each of the leases were assumed to be fully utilised at any one time (i.e. the entire 540 ha available in Oyster Harbour).
• The contribution of restored flat oyster (*Ostrea angasi*) reefs in OH was considered negligible given they occupy less 10 ha.

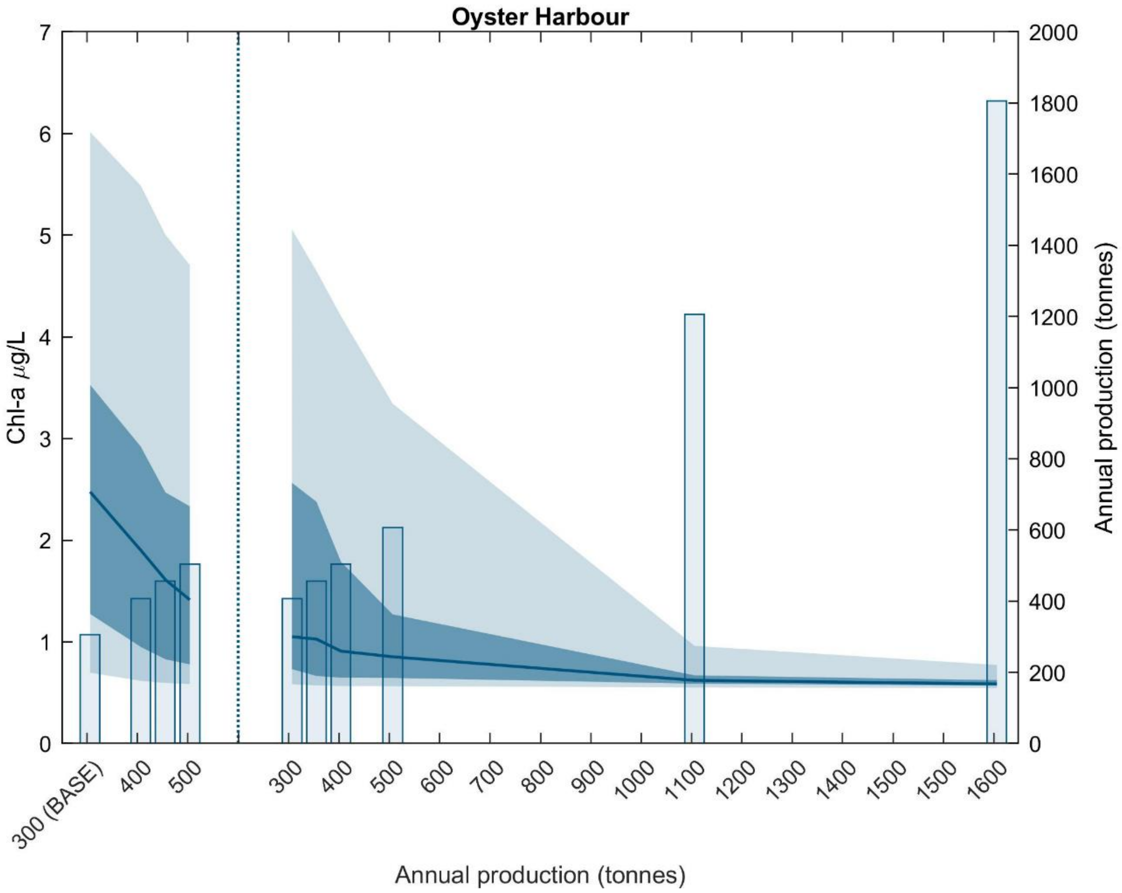
5.3.10 Results

Summary results are presented individually for each area. Figure 5.6 to Figure 5.8 present the depletion of phytoplankton against simulated annual productions, while Table 5.9 to Table 5.11 summarise the variations in nutrient depletion between scenarios. The results focus on changes in nutrient levels in OH and the subsequent impacts on nutrient levels in PRH and KGS.

Simulated time series for each of the modelled areas are presented in Appendix A. Each figure presents the modelled water quality in response to the differing production scenarios compared against the baseline scenario. Values are given for both surface and bottom indicators.

5.3.10.1 Oyster Harbour

Figure 5.6 presents phytoplankton volume (as chlorophyll-a) in response to increasing production levels, starting from the estimated current production level of 306 tonnes. Chlorophyll-a is presented as the shaded blue areas and the annual production values as histograms. Phytoplankton biomass declined with increasing production, even at relatively small annual production values (100-200 tonnes beyond the current baseline). At annual productions above 606 tonnes, phytoplankton volumes were consistently reduced to less than 1 µg.chlorophyll-a/L. Both particulate and dissolved organic carbon, the other two main food sources for shellfish, also show marked depletions with increasing shellfish production (Table 5.9).



Notes:
1. Each bar represents a different annual production scenario (from 1 to 3 in the left hand panel and 1 to 6 in the right hand panel, as in Table 5.7).
2. BASE = estimated production based on the standing biomass of farmed shellfish in OH presently.
3. The blue shaded areas represent the uncertainty in simulated clearance rates. The vertical extent of light blue shading from baseline to high annual production represents the 20th percentile of phytoplankton concentrations at the maximum clearance rate, to the 80th percentile of phytoplankton concentrations at the minimum clearance rate. The medium blue shading represents the median phytoplankton concentrations at the maximum clearance to median phytoplankton concentrations at the minimum clearance rate. The solid blue line represents the simulated median phytoplankton concentration at the medium clearance rate.

4. The bars are separated into two sections. The left hand section represents phytoplankton depletion in Oyster Harbour with no farming in Princess Royal Harbour or King George Sound; the right hand section represents depletion in Oyster Harbour with 14-27 tonnes and 804–1111 tonnes in Princess Royal Harbour and King George Sound respectively.

Figure 5.6 Phytoplankton depletion in Oyster Harbour with increasing annual production of *S. glomerata*.

The modelled rate of food depletion is linked to shellfish clearance rates. For phytoplankton, the percentage depletions from median baseline values were 27.35%–82.34%, 35.30%–76.36% and 42.62%–55.34% for the low to high production targets and minimum, medium and maximum clearance rates, respectively. For POC, the depletions were <5%–10.70%, <5%–22.02% and 20.80%–30.89% for the minimum, medium and maximum clearance rates respectively. Percentage depletions may be lesser for higher production scenarios/clearance rates in comparison to lower production scenarios/clearance rates as they are compared directly to the relevant median baseline value for that scenario, calculated on the current standing biomass of farmed shellfish (see Table 5.9).

Phytoplankton drawdown in OH changed in response to the scale of farming activities in PRH or KGS (Table 5.9). For example, when shellfish were included in KGS (shellfish in PRH had no effect on food availability in OH), phytoplankton depletions for an annual production of 200 tonnes were up to 82.30%, 76.36% and 55.34% for minimum, medium and maximum clearance rates respectively as opposed to 33.90%, 59.95% and 39.01% when shellfish were excluded; representing a difference between 16-48%. Similar results, though to a more significant extent, were demonstrated for POC. The results highlight the importance of considering the SCADZ as an inter-connected system, rather than three individual systems (see also Figure 5.7 and Figure 5.8).

Phytoplankton concentrations in OH range naturally between 0.2–11 µg.chlorophyll-a/L (Hillman 1991, Brearley 2005). At these levels, OH maintains phytoplankton volumes higher than typical oligotrophic systems in the region (~1.0 µg.chlorophyll-a/L) (Brearley 2005). Phytoplankton recharge is achieved to some extent via its connectivity to KGS, but mainly via seasonal inflows from the King and Kalgan rivers (Brearley 2005). Winter inflows carrying excessive agricultural inputs have contributed to the proliferation of epiphytes and macroalgae, smothering local seagrass communities (Brearley 2005). OH is therefore significantly modified from its baseline condition prior to European settlement.

According to the model, 506 to 606 tonnes annual production reduced chlorophyll-a concentrations to between 0.64 and 1.7 µg.chlorophyll-a/L, based on maximum and minimum simulated clearance rates respectively (Figure 5.6). Higher production values yielded chlorophyll-a concentrations consistently below 1 µg.chlorophyll-a/L, irrespective of the clearance rate simulated (see also Appendix A).

Table 5.9 Summary of nutrient depletions across varying shellfish productions for a given clearance rate in Oyster Harbour

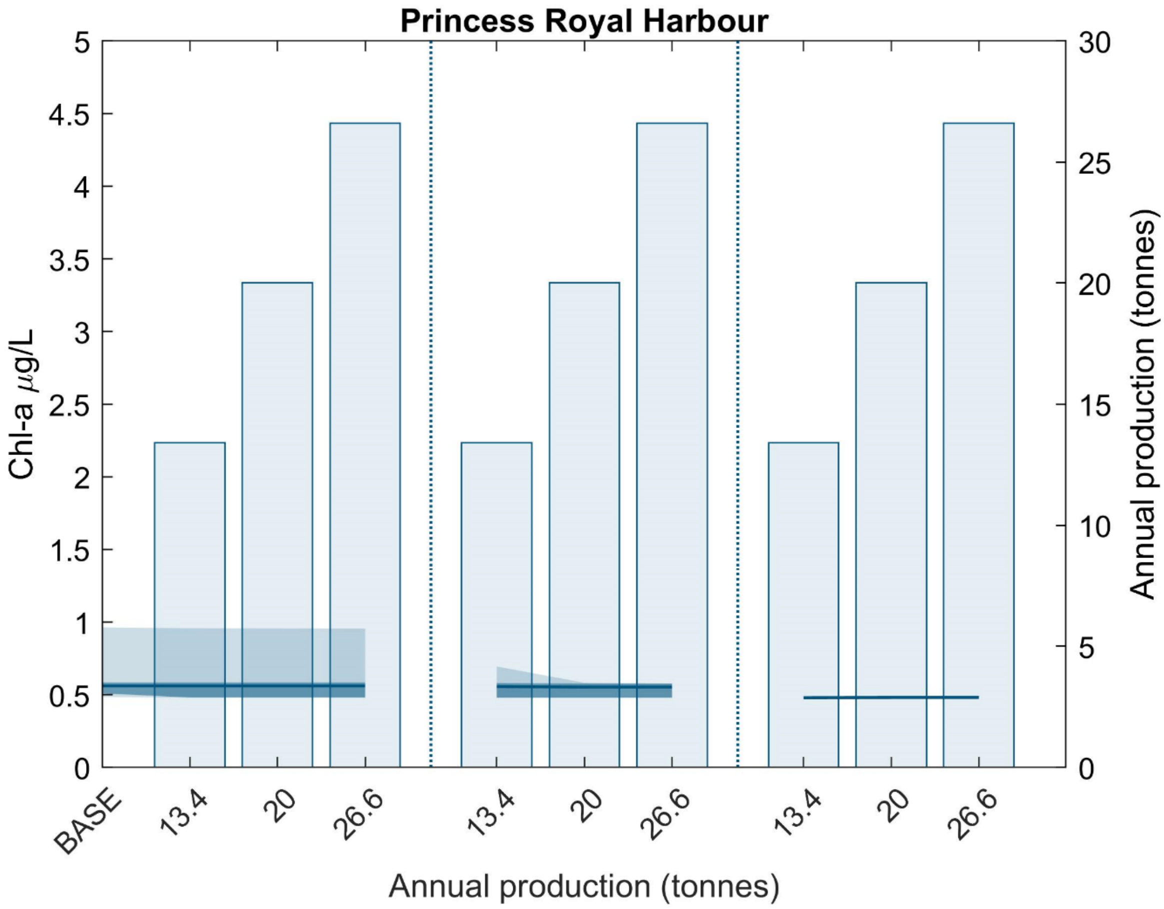
Standing biomass (tonnes)[1]	Annual production (tonnes)	Median chlorophyll-a concentration (µg/L) and % depletion from baseline		Median POC concentration (mg/L) and % depletion from baseline		Clearance rate
918 (base)[2]	306 (base)	3.53		0.33		1.3 L/hr
1218	406	2.56	27.35	0.32	<5[3]	
1368	456	2.38	32.63	0.32	<5[3]	
1518	506	1.79	49.30	0.31	<5[3]	
1818	606	1.27	64.03	0.30	7.65	
3618	1206	0.67	81.01	0.29	10.09	
5418	1806	0.62	82.34	0.29	10.70	
918 (base)[2]	306 (base)	2.48		0.34		2.2 L/hr
1218	406	1.63	35.30	0.32	<5[3]	
1368	456	1.37	44.80	0.31	9.00	
1518	506	1.16	53.30	0.31	9.00	
1818	606	1.05	57.50	0.30	11.76	
3618	1206	0.62	74.95	0.26	22.02	
5418	1806	0.59	76.36	0.28	18.16	
918 (base)[2]	306 (base)	1.27		0.28		4 L/hr
1218	406	0.73	42.62	0.26	20.80	
1368	456	0.66	47.96	0.25	23.55	
1518	506	0.65	49.22	0.24	25.38	
1818	606	0.64	49.45	0.25	25.08	
3618	1206	0.59	53.93	0.23	30.58	
5418	1806	0.57	55.34	0.23	30.89	
Results with only baseline biomasses present in King George Sound (with zero biomass in PRH)[4]						
918 (base)[2]	306 (base)	3.53		0.33		1.3 L/hr
1218	406	2.92	17.29	0.33	<5[3]	
1368	456	2.47	29.99	0.32	<5[3]	
1518	506	2.33	33.90	0.32	<5[3]	
918 (base)[2]	306 (base)	2.48		0.34		2.2 L/hr
1218	406	1.90	46.17	0.33	<5[3]	
1368	456	1.61	54.34	0.32	<5[3]	
1518	506	1.41	59.95	0.31	6.85	
918 (base)[2]	306 (base)	1.27		0.28		4 L/hr
1218	406	0.95	25.51	0.28	<5[3]	
1368	456	0.83	35.01	0.26	7.07	
1518	506	0.78	39.01	0.26	8.13	

Notes.
1. Standing biomasses of farmed shellfish. Biofouling biomasses are modelled at 50% of farmed shellfish biomasses.
2. Baseline biomasses were provided by industry.
3. Depletions of <5% are within model uncertainty and as such do not represent significant depletions.
4. Only three productions for Oyster Harbour were modelled with baseline shellfish in King George Sound and Princess Royal Harbour.

5.3.10.2 Princess Royal Harbour

Figure 5.6 illustrates the modelled response of phytoplankton in PRH to (a) increasing production in PRH (14 to 27 tonnes) assuming no production in OH or KGS (b) the response in PRH assuming 406–506 tonnes and 804–1111 tonnes production in OH and KGS respectively; and (c) the response in PRH assuming 606–1806 tonnes and 804–1111 tonnes in OH and KGS, respectively.

As in Figure 5.6, the shaded blue areas represent phytoplankton depletion and its associated uncertainty with differing clearance rates. The modelled results illustrate the level of connectivity between OH, KGS and PRH. Even at relatively minor production in KGS and OH, phytoplankton depletion in PRH increases (Figure 5.7). As summarised in Table 5.10, both particulate and dissolved organic carbon show marked depletions with increasing production, both when considering PRH alone, and with the addition of OH and KGS.



Notes:
1. Each bar represents a different annual production scenario for Princess Royal Harbour (from 1 to 3 as in Table 5.7).
2. BASE = estimated production based on the standing biomass of farmed shellfish in PRH presently, which was equivalent to zero tonnes.
3. The blue shaded areas represent the uncertainty in simulated clearance rates and seasonal variability. The vertical extent of light blue shading at baseline (i.e. zero annual production so no clearance) represents the 20th to 80th percentile of simulated phytoplankton concentrations as a proxy for natural variation. For scenarios when shellfish production is present, the light blue shading represents the 20th percentile at the maximum clearance rate to the 80th percentile at the minimum clearance rate. The medium blue shading represents the median phytoplankton concentrations at the maximum clearance to median phytoplankton concentrations at the minimum clearance rate. The solid blue line represents the simulated median phytoplankton concentration at the medium clearance rate.
4. The bars are separated into three sections. The left hand section represents phytoplankton depletion in Princess Royal Harbour when no shellfish are present in Oyster Harbour or King George Sound; the central section represents depletion in Princess Royal Harbour when 406–506 tonnes and 804–1111 tonnes (including estimated current productions of farmed shellfish) is simulated in Oyster Harbour and King George Sound respectively; and the right hand section represents depletion in Princess Royal Harbour when 606–1806 tonnes and 804–1111 tonnes (including estimated current productions of farmed shellfish) is simulated in Oyster Harbour and King George Sound respectively.

Figure 5.7 Phytoplankton depletion in Princess Royal Harbour with increasing annual production of *S. glomerata*, dependent on increasing annual production in Oyster Harbour and sustained annual production in King George Sound

The rate of food depletion in PRH was linked to shellfish clearance rates, however, as the modelled production numbers were very small the rate of depletion was also small. For phytoplankton, the modelled depletions were all less than 5% except under the maximum clearance scenario, where phytoplankton was depleted by 5.9%. For POC, depletions were <5%–16.66% for the minimum and medium clearance rates respectively, and 38.98% for the maximum clearance rate.

The modelled response of PRH, as phytoplankton concentrations are relatively low, any biomass of shellfish will potentially reduce phytoplankton concentrations to baseline levels (Table 5.10). Results also suggest that the connectivity between PRH and KGS (and by extension OH) may impact phytoplankton and POC levels in PRH to relatively minor degree. For example, POC depletions based on 27 tonnes annual production were between 16.66-38.98% when shellfish were included in OH and KGS, but were <5% when shellfish were excluded; representing a maximum difference of ~33%.

Historical records suggest phytoplankton concentrations in PRH range between 0.3–3 µg.chlorophyll-a/L (Hillman 1991, Brearley 2005). PRH like OH is a marine embayment with direct connectivity to KGS, with the concentration of phytoplankton (currently ~0.5 µg.chlorophyll-a/L with seasonal variation) primarily maintained through recharge with KGS. Though no major rivers flow into PRH, a network of drains collects water from agricultural and industrial land in the surrounding areas. These flow directly into PRH where they have historically contributed to year around elevated nutrient concentrations (Brearley 2005). As such, though the addition of shellfish into PRH could help further improve water quality, significant reductions in nutrient concentrations have already been achieved in PRH due to reduced urban and agricultural runoff.

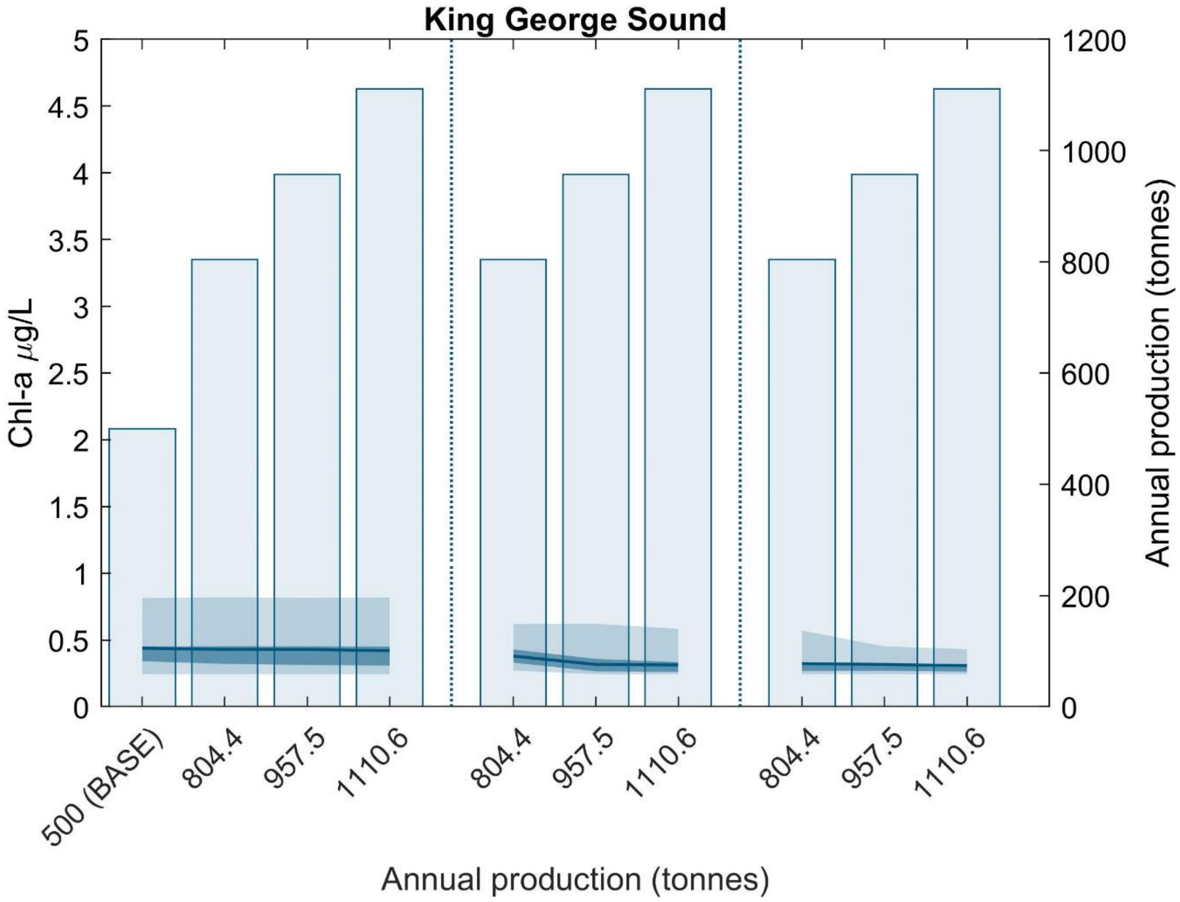
Table 5.10 Summary of nutrient depletions across varying *S. glomerata* productions for a given clearance rate in Princess Royal Harbour, dependent on increasing annual production in Oyster Harbour and King George Sound

Standing biomass (tonnes)[1]	Annual production (tonnes)	Oyster Harbour Annual Production[2]	King George Sound Annual Production[2]	Median chlorophyll concentration (µg/L) and % depletion from baseline		Median POC concentration (mg/L) and % depletion from baseline		Clearance rate[3]
0 (PRH base)		306 (base)	500 (base)	0.58		0.06		1.3 L/hr
40	14	406	804	0.58	<5[4]	0.06	<5[4]	
60	20	456	958	0.58	<5[4]	0.05	16.66	
80	27	506	1111	0.58	<5[4]	0.05	16.66	
0 (PRH base)		306 (base)	500 (base)	0.56		0.06		2.2 L/hr
40	14	406	804	0.56	<5[4]	0.06	<5[4]	
60	20	456	958	0.56	<5[4]	0.05	16.66	
80	27	506	1111	0.56	<5[4]	0.05	16.66	
0 (PRH base)		306 (base)	500 (base)	0.51		0.06		4 L/hr
40	14	406	804	0.48	5.90	0.04	38.98	
60	20	456	958	0.48	5.90	0.04	38.98	
80	27	506	1111	0.48	5.90	0.05	17.33	
Results with only baseline biomasses present in Oyster Harbour King George Sound[2]								
0 (PRH base)		306 (base)	500 (base)	0.58		0.06		1.3 L/hr
40	14			0.58	<5[4]	0.06	<5[4]	
60	20			0.58	<5[4]	0.06	<5[4]	
80	27			0.58	<5[4]	0.06	<5[4]	
0 (PRH base)		306 (base)	500 (base)	0.56		0.06		2.2 L/hr
40	14			0.56	<5[4]	0.06	<5[4]	
60	20			0.56	<5[4]	0.06	<5[4]	
80	27			0.56	<5[4]	0.06	<5[4]	
0 (PRH base)		306 (base)	500 (base)	0.51		0.06		4 L/hr
40	14			0.48	5.90	0.06	<5[4]	
60	20			0.48	5.90	0.06	<5[4]	
80	27			0.48	5.90	0.06	<5[4]	

Notes.
1. Standing biomasses of farmed shellfish. Biofouling biomasses are modelled at 50% of farmed shellfish biomasses.
2. Only three production targets for Oyster Harbour/King George Sound were modelled for the depletion effects in Princess Royal Harbour.
3. Clearance rates are for *S. glomerata*. Shellfish present in King George Sound clear at rates detailed in Table 5.7
4. Depletions of <5% are within model uncertainty and as such do not represent significant depletions

5.3.10.3 King George Sound

Figure 5.8 presents phytoplankton depletion as a function of shellfish production in KGS, based on 68 to 139 tonnes for *P. fucata*, and 736 to 972 tonnes for *M. galloprovincialis*. Depletions are represented with and without farming in Oyster and Princess Royal Harbours. The left hand panel represents phytoplankton depletion in KGS assuming no shellfish in OH or PRH; the central panel represents the effect on KGS following an increase in production from 406–506 tonnes in OH and 13–27 tonnes in PRH; and the right hand panel represents the effect on KGS following an increases in production from 606–1806 tonnes in OH, with 13–27 tonnes production in PRH. As in Figure 5.6, the shaded blue areas represent phytoplankton depletion and its associated uncertainty with differing clearance rates. As summarised in Table 5.11, both particulate and dissolved organic carbon also show significant depletions as shellfish productions increase.



Notes:

1. Each bar represents a different annual production scenario for King George Sound (from 1 to 3 as in Table 5.7; including production of both P. fucata and M. galloprovincialis).
2. BASE = estimated production based on the standing biomass of farmed shellfish in KGS presently.
3. The blue shaded areas represent the uncertainty in simulated clearance rates. The vertical extent of light blue shading from baseline to high annual production represents the 20th percentile of phytoplankton concentrations at the maximum clearance rate to the 80th percentile of phytoplankton concentrations at the minimum clearance rate. The medium blue shading represents the median phytoplankton concentrations at the maximum clearance to median phytoplankton concentrations at the minimum clearance rate. The solid blue line represents the simulated median phytoplankton concentration at the medium clearance rate.
4. The bars are separated into three sections. The left hand section represents phytoplankton depletion in King George Sound when no shellfish are present in Oyster Harbour or Princess Royal Harbour; the central section represents depletion in King George Sound when 406–506 tonnes (including estimated current productions of farmed shellfish) and 13-27 tonnes is simulated in Oyster Harbour and Princess Royal Harbour respectively; and the right hand section represents depletion in King George Sound when 606–1806 tonnes (including estimated current productions of farmed shellfish) and 13-27 tonnes is simulated in Oyster Harbour and Princess Royal Harbour respectively.

Figure 5.8 Phytoplankton depletion in King George Sound with increasing annual production of *P. fucata* and *M. galloprovincialis*, dependent on increasing annual production in Oyster Harbour and sustained annual production in Princess Royal Harbour

Unlike OH and PRH, KGS has remained relatively pristine with records pointing to a typically oligotrophic system between 0.1–1.1 µg.chlorophyll-a/L (Hillman 1991, DWER 2020). The level of food depletion was directly related to shellfish clearance rates. Depletions from baseline ranged between <5%–26.11%, 8.22%–28.54% and <5%–23.46% for minimum, medium and maximum clearance rates respectively. For POC, depletions ranged between 7.70%–15.38% for minimum and medium clearance rates, and between 8.40%–9.24% for maximum clearance rates.

As in the other examples, results illustrate the effect of farming in OH (and PRH though to a minor degree) on food availability in KGS. Concurrent farming in OH and PRH resulted in at least a 12% decline in food availability in KGS. Similar results were reported for POC, again reinforcing the importance of considering shellfish production targets, and associated carrying capacities, in all areas of the SCADZ in conjunction rather than independently.

Table 5.11 Summary of nutrient depletions across varying *P. fucata* and *M. galloprovincialis* productions for a given clearance rate in King George Sound, dependent on increasing annual production in Oyster Harbour

Standing biomass (tonnes)[1]	Annual production (tonnes)	Oyster Harbour Annual production (tonnes)[2]	Median chlorophyll concentration (µg/L) and % depletion from baseline		Median POC concentration (mg/L) and % depletion from baseline		Clearance rate
500[5] (base)	500 (base)	306 (base)	0.45		0.13		5 L/hr[4] 1.3 L/hr[5]
68[4]	68	406	0.43	<5[6]	0.11	15.38	
736[5]	736						
104[4]	104	456	0.36	20.80	0.12	7.70	
854[5]	854						
139[4]	139	506	0.33	26.11	0.12	7.70	
972[5]	972						
500[5] (base)	500 (base)	306 (base)	0.44		0.13		15 L/hr[4] 2 L/hr[5]
68[4]	68	406	0.40	8.22	0.11	15.38	
736[5]	736						
104[4]	104	456	0.32	27.63	0.12	7.70	
854[5]	854						
139[4]	139	506	0.31	28.54	0.12	7.70	
972[5]	972						
500[5] (base)	500 (base)	306 (base)	0.34		0.12		30 L/hr[4] 4 L/hr[5]
68[4]	68	406	0.33	<5[6]	0.11	8.40	
736[5]	736						
104[4]	104	456	0.26	22.58	0.11	8.40	
854[5]	854						
139[4]	139	506	0.26	23.46	0.11	9.24	
972[5]	972						
Only baseline shellfish biomasses present in Oyster Harbour (PRH has a base of 0)							
500[5] (base)	500 (base)	306 (base)	0.45		0.13		5 L/hr[4] 1.3 L/hr[5]
68[4]	68		0.45	<5[6]	0.13	<5[6]	
736[5]	736						
104[4]	104		0.45	<5[6]	0.13	<5[6]	
854[5]	854						
139[4]	139		0.45	<5[6]	0.13	<5[6]	
972[5]	972						
500[5] (base)	500 (base)	306 (base)	0.44		0.13		15 L/hr[4] 2 L/hr[5]
68[4]	68		0.43	<5[6]	0.13	<5[6]	
736[5]	736						
104[4]	104		0.43	<5[6]	0.13	<5[6]	
854[5]	854						
139[4]	139		0.42	<5[6]	0.13	<5[6]	
972[5]	972						
500[5] (base)	500 (base)	306 (base)	0.34		0.12		30 L/hr[4] 4 L/hr[5]
68[4]	68		0.32	5.86	0.12	<5[6]	
736[5]	736						
104[4]	104		0.31	8.21	0.12	<5[6]	
854[5]	854						
139[4]	139		0.31	9.97	0.12	<5[6]	
972[5]	972						

Notes:
1. Standing biomasses of farmed shellfish. Biofouling biomasses are modelled at 50% of farmed shellfish biomasses.
2. Only three production targets for Oyster Harbour were modelled for the depletion effects in King George Sound.
3. Baseline biomasses were provided by industry.
4. Biomasses/clearance rates for *P. fucata*.
5. Biomasses/clearance rates for *M. galloprovincialis.*
6. Depletions of <5% are within model uncertainty and as such do not represent significant depletions

6 Benthic nutrient enrichment

6.1 Overview of studies

Benthic nutrient enrichment refers to the increased nutrient inputs from aquaculture operations that settle on the seabed. Benthic nutrient enrichment may occur due to increased nutrients from feed inputs and bio-deposition of faecal matter (Joyce, Rubio and Winberg 2010, Hargrave et al. 2008). The risk of benthic nutrient enrichment for shellfish aquaculture is lower than that of finfish as feed inputs are not involved, meaning in many cases no nutrient enrichment from shellfish aquaculture may occur (Crawford et al. 2003).

While shellfish bio-deposition is an important process in areas where shellfish are naturally found, allowing for remineralisation of organic matter by the microbial and benthic fauna community (Grant et al. 2005), it may lead to unhealthy organic enrichment if excess nutrient loading occurs beyond normal levels in an ecosystem. For example, increased sediment nutrient content may allow opportunistic functional groups (generally macroalgae) to outcompete other ecologically important functional groups that were originally dominant (Callier et al. 2010; Liao et al. 2019). A change in the nutrient content of the sediments can also be cycled through to the rest of the ecosystem (Carlsson et al. 2012), leading to trophic cascades whereby other organisms (such as phytoplankton) may proliferate. Further, increased sedimentation from bio-deposition is also common, which can smother benthic habitats present underneath aquaculture infrastructure (Dahlback and Gunnarsson 1981; Grant et al. 2005; Mitchell 2005). Finally, the breakdown of bio-deposits can lead to increased sediment oxygen uptake and as such sediment anoxia and the accumulation of free sulphide (Dahlback and Gunnarsson 1981).

Much of the risk of benthic nutrient enrichment from shellfish aquaculture depends on the hydrodynamics of the region in which the shellfish are grown (Chamberlain et al. 2001, Hayakawa et al. 2001). For example, in an area where there is considerable flushing of the waterway where bio-deposited material is dispersed away from the aquaculture infrastructure and not concentrated, then the risk of nutrient enrichment directly underneath the infrastructure is low (Mitchell 2005). However, in an area where there is low flushing and low hydrodynamic movement, the risk of nutrient enrichment is much higher even if the rate of bio-deposition itself from the shellfish is low (Grant et al. 2005).

6.2 Potential impacts

The potential impacts to benthic communities are outlined in Table 6.1.

Table 6.1 Potential direct and indirect impacts to marine environmental quality

Potential impacts	Context
Direct impacts	• Aquaculture infrastructure (posts, anchors, lines) directly obstructing / reducing regular water currents / flows • Aquaculture infrastructure / vessel movements disturbing sediments and releasing particulate matter into the water column
Indirect impacts	• Deposition of faeces and pseudofaeces from shellfish leading to benthic nutrient enrichment beyond natural levels • Deposition of biofouling from aquaculture infrastructure directly leading to benthic nutrient enrichment beyond natural levels • Shellfish filter-feeding draw down on particulate matter and organisms in the water column (see Section 5.2)

6.3 Methods

The risk of benthic nutrient enrichment from shellfish bio-deposition was calculated by multiplying the bio-deposition rate by the standing biomass estimated under each scenario for the carrying capacity section.

The potential for benthic nutrient enrichment was considered in the context of the modelled bio-deposition rates compared to the rates reported in the literature (Table 6.2). The intent of the review was to identify the rates of bio-deposition that may lead to changes in sediment chemistry, relative to the rates predicted by modelling.

Table 6.2 Comparison of bio-deposition results with the results published in the literature

Area	Species	Bio-deposition rate (g [dry weight]/m²/day)	Retention time (days)	Current speed (m/s)	Impact assessment	References
Oyster Harbour	Sydney rock oyster (*S. glomerata*)	1.8–8.027	12.5	-	Potential for minor enrichment within lease area	N/A
Princess Royal Harbour	Sydney rock oyster (*S. glomerata*)	0.536–1.06	12.5	-	Potential for minor enrichment within lease area	N/A
King George Sound	Akoya pearl oysters (*P. fucata*)	0.608–1.23	30	-	Potential for minor enrichment within lease area	N/A
King George Sound	Blue mussels (*M. galloprovincialis*)	4.264–5.864	30	-	Potential for minor enrichment within lease area	N/A
Southern Tasmania	Pacific oyster (*Crassostrea gigas*)	180.5–39.6	1.5	-	No impact	Mitchell 2005
Southern Tasmania	Pacific oyster (*C. gigas*)	14.5	-	0.034–0.18	No impact	Crawford et al. 2003
Nova Scotia, Canada	Blue mussels (*M. edulis*)	88.9	15	-	No impact	Grant et al. 1995
Ofunato Estuary, Japan	Pacific oyster (*C. gigas*)	10–80	-	0.15	No impact	Hayakawa et al. 2001
Southwest Ireland	Blue mussels (*M. edulis*)	-		0.023–0.034	Minor change restricted to within 40m of farm	Chamberlain et al. 2001
Swedish west coast	Blue mussels (*M. edulis*)	2.4–3.3 (g [dry weight] carbon /m² /day)	-	0.03	Increased carbon content under aquaculture relative to reference sites	Dahlback and Gunnarsson 1981
Swedish west coast	Blue mussels (*M. edulis*)	28–36 impact 19–23 reference	-	-	Increased sedimentation and nitrogen flux under aquaculture relative to reference sites	Carlsson et al. 2012

Area	Species	Bio-deposition rate (g [dry weight]/m^2/day)	Retention time (days)	Current speed (m/s)	Impact assessment	References
Bay of Morlais (France)	Pacific oyster (*C. gigas*)	0.066–0.246	-	-	Minimal to no impact on sediments or fauna	Boucher & Boucher-Rodoni 1988

Notes
1. Bio-deposition resulting in no material impact (green highlight); bio-deposition resulting in material impact (red highlight)

6.4 Results

Rates of bio-deposition leading to sediment nutrient enrichment varied between studies. Some studies recorded no material impacts to sediments at bio-deposition rates of up to 180.5 grams [dry weight] per m^2 per day (Mitchell 2005), while others recorded increased sedimentation and nitrogen fluxes in sediments with bio-deposition rates of 28 grams [dry weight] per m^2 per day (Carlsson et al. 2012).

These rates of bio-deposition are an order of magnitude higher than those predicted in this study for PRH and KGS, which were between 0.536–1.06 and 0.608–5.86 grams [dry weight] per m^2 per day respectively.

The maximum risk recorded in this study relates to the farming of *S. glomerata* in OH, which according to the model, could achieve a maximum bio-deposition rate of 8.027 grams [dry weight] per m^2 per day; a figure which is at least an order of magnitude lower than the bio-deposition rates that caused minor impacts in other studies (Carlsson et al. 2012).

Based on these data, it is considered that the proposed farming in the SCADZ poses a very low risk to the benthic environment, and that any bio-deposition will be constrained to within areas immediately below the aquaculture infrastructure with no impacts to adjacent sediments. Any risks posed by the accumulation of faecal waste are likely to be mitigated in the winter months when increased storm activity and river flow is likely to contribute to the resuspension of organic materials, and/or periodic resetting events, with the effect of returning sediment quality to baseline levels.

7 Harmful Algal Blooms

7.1 Overview of studies

Harmful algal blooms, or HABs, are blooms of toxic or harmful species (Anderson et al. 2012). HABs are common in coastal marine as well as freshwater and brackish ecosystems. Generally, they are caused by blooms of noxious or toxic microscopic algae or cyanobacteria (blue-green algae), which either proliferate or build biomass rapidly at the water surface or in the water column under suitable conditions.

HABs occur regularly in Australian coastal environments, posing risks to aquaculture operations. Monitoring in south-eastern Australia found concentrations of harmful species, particularly *Dinophysis* sp., have increased in the last decade (Brett et al. 2020). In Western Australia, the Department of Health has consistently monitored the occurrence of HABs in areas of concern for shellfish, including Albany, under the West Australian Shellfish Quality Assurance Program (DoH 2017) and Marine Biotoxin Monitoring and Management Program (DoH 2016). HABs are also monitored in Oyster Harbour by the Department of Water and Environmental Regulation as part of the estuaries' initiative (Thomson 2018).

Albany waters' support various HABs, including diatoms, dinoflagellates and cyanobacteria. HAB concentrations occasionally exceed health recommendations outlined by DWER. The guideline value for *Dinophysis acuminata* has been equalled or exceeded four times between 2016 and 2017. Princess Royal Harbour (PRH) has experienced two harmful algal bloom events (concentrations above flesh testing alert level) since 2017, one of *Dinophysis acuminata* and one of *Karenia brevis* (DPIRD pers. comm). HABs (*Prorocentrum rhathymum*) have also been recorded at Mistaken Island, where in 2017 maximum concentrations of 330 cells per L were recorded (DPIRD pers. comm).

HAB cysts, particularly dinoflagellates, may lie dormant in marine sediments for several years (Anderson et al. 2012). Disturbance of sediments through the placement of infrastructure may dislodge the cysts, posing a risk to local aquaculture. Cysts may also be released by wind driven wave action, which could result in the movement of HAB cysts to other areas.

The risk posed by HABs may increase with the effects of climate change (O'Neil et al. 2012, Wetz & Yoskowitz 2013, Philips et al. 2020). For example, high levels of domoic acid toxin occurred in oysters in Wogonga Lake, NSW in 2016 following a diatom bloom of several *Pseudonitzschia* spp. These events were associated with extreme bushfires followed by flooding, resulting in unprecedented levels of toxins responsible for amnesic shellfish poisoning (ASP) (DPIRD pers. comm).

Altered nutrient ratios either as a result of climate change or other external factors may also influence the toxicity of HABs. For example, altered phosphate levels compared to nitrate were found to increase toxicity in the Prymnesiophyte *Chrysochromulina polylepis* (DPIRD pers. comm). In Albany, longer term dry conditions and reduced river flows may lead to changes in several parameters, all of which could alter the toxicity of HAB groups already found within the region.

7.2 Potential impacts

Several types of HABs are of direct concern to aquaculture developments and health authorities, as toxic species can accumulate in the flesh of shellfish which filter-feed the algae from the water column. Though in many cases the bloom does not cause direct harm to the shellfish, it can result in severe illness and/or death in humans if the flesh is consumed, as detailed below:

- Amnesic Shellfish Poisoning (ASP)
  - Predominantly caused by *Pseudonitzschia* sp. (nutrient limited diatom)
  - Potentially fatal
- Ciguatera Fish Poisoning (CFP)

    - Not of concern in this study
- Diarrhoetic Shellfish Poisoning (DSP)
    - *Dinophysis* sp. (dinoflagellate – not nutrient limited)
    - Not fatal
- Neurotoxic Shellfish Poisoning (NSP)
    - *Karenia brevis* (dinoflagellate)
    - Not fatal
- Paralytic Shellfish Poisoning (PSP)
    - *Alexandrium* spp., *Gymnodinium catenatum*, *Pyrodinium bahamense* (all dinoflagellates)
    - Potentially fatal.

The potential impacts of HABs in relation to human health are described in Table 7.1.

Table 7.1 Most likely direct and indirect impacts of algal blooms on human health

Potential impacts	Context
Direct impacts	• Some species cause major irritation and damage to the skin of organisms, which may in certain scenarios be fatal. • Illness in humans may occur when toxic species are ingested by organisms either directly or bio-accumulated through the food web. Poisoning following ingestion may in some circumstances lead to human fatalities.
Secondary & tertiary impacts	• Oxygen drawdown in the water column following the death and decomposition of HABs. • Creation of noxious scum or foam.

7.3 Approach to assessment

The extent to which the expansion of the shellfish industry may lead to a higher risk of HABs was investigated using a first principals' approach.

7.4 Methods

7.4.1 Model

Risks were assessed based on changes in modelled water quality under a range of scenarios. For further details, refer to Section 5.3.8.

7.4.2 Adaptation for risk assessment of harmful algal blooms

The assessment proceeded on the assumption that if the proposed expansion resulted in significant changes to the local nutrient budget, then the risk of HABs may also change relative to the status quo. The assessment focussed on nutrients, but also salinity, temperature and dissolved oxygen.

7.5 Results

Figure 5.6 to Figure 5.8, as well as the timeseries in Appendix A, show the modelled results for a range of water quality parameters, under the full range of scenarios. Total organic carbon (in the form of POC and DOC) showed depletions across the modelled domains, but particularly in OH and KGS. Further analysis of the time series data (Appendix A), found a reduction in total phosphorous (from baseline) of up to 75%, marginal decline in total nitrogen and no changes in sulphur. Other environmental parameters with the potential to influence algal growth, such as dissolved oxygen, silicate, temperature and salinity, were also unaffected.

Modelling indicated that with the introduction of greater shellfish biomasses the risk of HABs in each of these areas is unlikely to change, if not decrease, particularly given the predicted reductions in bioavailable phosphorus. While this may have the effect of decreasing the incidences and abundances of nutrient limited guilds of HABs, risks associated with dinoflagellates (which are affected more by changes in salinity and temperature (Kamiyama et al. 2010; Ajani et al. 2016) are not expected to deviate from the status quo.

The risks posed by HABs is therefore likely to remain moderate (conditionally approved) in OH and low (approved) in KGS as determined under previous WASQAP classifications. The level of risk in PRH is pending further study, but the results are pointing to a moderate risk.

8 Conclusions

8.1 Benthic communities and habitats assessment - Albany

The potential for permanent (>5 years recovery) and/or recoverable losses (<5 years recovery) of BCH was calculated based on the most likely positioning and configuration of shellfish farming infrastructure (i.e. posts, longlines and baskets). Irrecoverable losses in Albany were shown to be negligible at <0.1% and recoverable losses minimal at <5%. The ecological function of BCH in Albany is therefore unlikely to be impacted by farming operations, if best practice operations and management strategies are followed.

8.2 Benthic communities and habitats assessment - Esperance

The potential for permanent and/or recoverable losses of BCH in Esperance was considered in the context of abalone ranching. The impact potential was negligible because the artificial habitats used for ranching (ABITATs) are placed exclusively on sandy substrates, at a distance of at least 50 m from seagrasses or macroalgal communities. As such, no calculations of future losses attributable to the SCADZ were undertaken for Esperance. The proposed expansion of the SCADZ to Esperance therefore poses no risk to the ecological function of BCH in the local region.

8.3 Carrying capacity

Phytoplankton biomass and suspended organic material concentrations declined significantly with increasing shellfish production. Food depletion in one water body was linked to farming activities in adjoining water bodies. For example, concurrent operations in KGS (and to a lesser extent PRH), increased food depletion in OH by at least 16% thus illustrating the importance of considering the SCADZ as an inter-connected system, rather than three individual systems. Based on the results, it is recommended that OH, KGS and PRH are managed as one system, rather than three independent systems.

The observed interconnectivity between OH, PRH and KGS may affect the capacity for food resupply both within and between the water bodies [Individually, each is recharged by inflows from rivers/drains, and together, because of the movement of water between them]. It is critical therefore that the volume of food cleared by shellfish does not exceed the recharge capacity of the individual water bodies or the combined system – ultimately affecting shellfish growth rates, and possibly, ecological function.

Typical healthy oligotrophic systems in the south west of Western Australia maintain phytoplankton concentrations (as indicated by chlorophyll-a) at around 1 µg.chlorophyll-a/L (Brearley 2005); whereas, according to ANZECC/ARMCANZ (2000), enriched ecosystems exceed 3 µg.chlorophyll-a/L. Historical data collected between 1988 and 2018 point to significant differences in the level of productivity between the OH, KGS and PRH water bodies. While KGS is clearly oligotrophic (0.05 and 1 µg.chlorophyll-a/L), OH and historically PRH regularly approach/ed or exceed/ed the ANZECC/ARMCANZ (2000) trigger for eutrophic systems, respectively); this is particularly the case for OH which has recorded concentrations as high as 11 µg.chlorophyll-a/L (Hillman 1991, Thomson 2018).

In this study, phytoplankton was depleted to <1 µg.chlorophyll-a/L based on annual productions of 506-606 t in OH, and <0.5 µg.chlorophyll-a/L based on annual productions of 20 t in PRH and 68 and 736 t in KGS, for *P. fucata* and *M. galloprovincialis* respectively. Phytoplankton concentrations are regularly <1 µg.chlorophyll-a/L in PRH and KGS, but not in OH (Thompson 2018, DWER 2020). Based on these results, modelled interim carrying capacity targets were extrapolated for each system. Targets presented below are inclusive of the standing biomasses currently farmed in the systems:

OH: **506 t** to **606 t**
PRH: **20 t**

KGS: **68 t** (*P. fucata*) and **736 t** (*M. galloprovincialis*)

The above targets do not include the contribution of biofouling, which was estimated to represent another 50% (equivalent weight) on top of the modelled production numbers. Since biofouling organisms compete with shellfish for food (phytoplankton and other organic material), there is an expectation that any reduction of biofouling achieved through farm management will reduce competition and in turn, allow for greater shellfish production.

The modelled targets for OH, KGS and PRH are considered safe and achievable based on the best available knowledge of regional food availability, with no expectation that they will lead to local ecological impacts. The interim targets serve as a conservative starting point for the safe expansion of the industry, pending collection of further data and ultimately, the reassessment of the carrying capacity targets. It is recommended that any expansion of the industry beyond the numbers presented herein, is undertaken with caution, and following the analysis of environmental monitoring and shellfish production data, together with the validation of the ecosystem model and its assumptions.

8.4 Risk of benthic nutrient enrichment

The potential for benthic nutrient enrichment was considered using the modelled bio-deposition rates coupled to the carrying capacity assessment. Modelled bio-deposition rates were considered in the context of the relevant literature, with comparisons made to other shellfish bio-deposition studies.

Published impact thresholds were highly variable. Some studies recorded bio-deposition rates of up to 180.5 grams [dry weight] per m^2 per day yet no material impacts to sediments (Mitchell 2005), while others recorded increased sedimentation and nitrogen flux in sediments at bio-deposition rates of 28 grams [dry weight] per m^2 per day (Carlsson et al. 2012). The maximum risk recorded in this study related to the farming of *S. glomerata* in OH, which according to the model, achieves a maximum bio-deposition rate of 8.027 grams [dry weight] per m^2 per day; a figure at least an order of magnitude lower than the bio-deposition rates demonstrated to result in minor impacts to sediments. Current farm practices ensure the consistent movement of aquaculture infrastructure (e.g. oyster baskets), reducing the potential cumulative impacts on benthic environments through a process of fallowing.

These results notwithstanding, the assessment of benthic nutrient enrichment was restricted to modelled rates of faecal and pseudo-faecal deposition. Management of the SCADZ should also consider the contribution of biofouling which under current farm practices is detached and disposed of to the surrounding water. Based on this, it is recommended that sediment nutrient parameters are monitored for an initial period until it can be ascertained that the impacts of biofouling removal are benign, and not at risk of exceeding the environmental quality guidelines.

8.5 Risk of harmful algal blooms

The risks posed by harmful algal blooms (HABs) in OH and KGS are known to operators and regulated under WASQAP. Under WASQAP, stringent monitoring is undertaken to ensure the safety of shellfish farmed in OH. Risks in PRH have not yet been classified, though elevated concentrations of harmful algae have been recorded in the past.

The potential for changes to the risks profile were determined using a first principles approach, based on the potential for the SCADZ to effect changes to regional water quality, particularly the characteristics likely to increase the probability of a bloom.

The study focused on nutrient limited algal groups (e.g. *Pseudonitzschia* sp.) and dinoflagellates. Modelling suggested the risks posed by nutrient limited groups will likely decline, whereas risks posed by dinoflagellates would remain unchanged. Risks posed by HABs were predicted to remain 'moderate' (conditionally approved) in OH, and 'low' (approved) in KGS as determined under previous classifications. The risk to PRH is pending further study, though results to date point to a 'moderate' level of risk. It is recommended that the approach to managing the risks be outlined in the relevant Marine Environmental Management Plans for the Zone.

It was also noted (in the review) that HAB assemblages may be affected by climate change, especially as extreme weather and fire events become more prevalent. Recent events associated with the January 2020 bush fires, for example, led to changes in the biochemistry of harmful algal groups in eastern Australia, leading to enhanced toxicity. Further work is needed to better understand the risk posed by changing environmental conditions, and what they may mean for the shellfish industry as it evolves.

8.6 Recommendations

Results presented here suggested the SCADZ poses a negligible risk to the marine environment, and that any residual risks posed by HABs are manageable under the WASQAP framework. Nonetheless, there remains scope to improve understanding and therefore management of the SCADZ via the following actions:

- Finetune the model and carrying capacity estimates using operational data and improved knowledge of shellfish bio-energetics (e.g. clearance rates, bio-deposition rates);
- Closely monitor the system in the initial phases of operations, cross-check the results with model outcomes (and validate and remodel as appropriate);
- Develop an interim sediment monitoring program around the aquaculture leases to evaluate the potential for enrichment due to faecal and biofouling deposition;
- Quantify the presence and abundance of HAB cysts in the sediments (primarily dinoflagellates) to better understand the underlying risk of HAB occurrences; and finally,
- Examine the potential effects of climate change to the industry, including the extent to which risks may change in the future.

9 References

ACWA (2013) Environmental Code of Practice for The Sustainable Management of Western Australia's Mussel and Oyster Aquaculture Industries.

Ajani P, Larsson M, Rubio A, Bush S, Brett S, Farrell H (2016) Modelling bloom formation of the toxic dinoflagellates *Dinophysis acuminata* and *Dinophysis caudata* in a highly modified estuary, south eastern Australia. Estuarine, Coastal and Shelf Science 183: 95-106.

Anderson D, Cembella A, Hallegraeff G (2012) Progress in understanding Harmful Algal Blooms: Paradigm shifts and new technologies for research, monitoring, and management. Annual Review of Marine Science 4: 143-176.

APA (2013) Albany Port Authority Environmental Management Plan. Report Prepared by Albany Port Authority, Albany, Western Australia.

Arnold W, White M, Norris H, Berrigan M (2000) Hard clam (*Mercenaria* spp.) aquaculture in Florida, USA: geographic information system applications to lease site selection. Aquacultural Engineering 23: 203–231.

ASQAP (2019) Australian Shellfish Quality Assurance Program Manual. Australian Shellfish Quality Assurance Advisory Committee.

Bayne, BL (1999) Physiological components of growth differences between individual oysters (*Crassostrea gigas*) and a comparison with *Saccostrea commercialis*. Journal of Physiology Biochemistry and Zoology 72:705-713.

Bastyan G (1986) Distribution of seagrasses in Princess Royal Harbour and Oyster Harbour, on the Southern Coast of Western Australia. Prepared for Department of Conservation and Environment.

Bastyan G, Cambridge M (2008) Transplantation as a method for restoring the seagrass *Posidonia australis*. Estuarine, Coastal and Shelf Science 79: 289-299.

Bastyan G (2015) The seagrass distribution of Middleton Bay. Prepared for the City of Albany.

BMT (2020) SCADZ Comprehensive Classification Survey: Sampling & Analysis Plan. Report No. R-1757_00-1. Prepared for the Department of Primary Industries and Regional Development by BMT Commercial Australia Pty Ltd, March 2020.

BMT Oceanica (2014) Albany Port Authority Maintenance Dredging Program Environmental Impact Assessment. Prepared for Albany Port Authority.

BMT Oceanica (2016) Wylie Bay Habitat Map. Prepared for Ocean Grown Abalone.

BMT Western Australia (2018) Middleton Beach Artificial Surf Reef Environmental Impact Assessment. Prepared for the City of Albany.

Brearley A (2005) Ernest Hodgkin's Swanland: estuaries and coastal lagoons of Southwestern Australia. Published by the University of Western Australia Press.

Brett S, Davies C, Eriksen R, Richardson A (2020) Harmful Algal Blooms and the shellfish industry. Chapter 3.4 in State and Trends of Australia's Oceans Report. Integrated Marine Observer System.

CA (2019a) Marine Pests. Available at http://www.marinepests.gov.au/marine_pests [Accessed March 2020].

Callier M, Richard M, McKindsey C, Archambault P, Desrosier G (2009) Responses of benthic macrofauna and biogeochemical fluxes to various levels of mussel bio-deposition: An in situ ''benthocosm" experiment. Marine Pollution Bulletin 58: 1544-1553.

Cambridge M, Bastyan G, Walker D (2002) Recovery of *Posidonia* meadows in Oyster Harbour, Southwestern Australia. Bulletin of Marine Science 71 (3); 1279-1299.

Carlsson M, Engstrom P, Lindahl O, Ljungqvist L, Petersen J, Svanberg L, Holmer M (2012) Effects of mussel farms on the benthic nitrogen cycle on the Swedish west coast. Aquaculture Environment Interactions 2: 177-191.

Chamberlain J, Fernandes T, Read P, Nickell T, Davies I (2001) Impacts of bio-deposits from suspended mussel (*Mytilus edulis* L.) culture on the surrounding surficial sediments. ICES Journal of Marine Science 58: 411-416.

Chamberlain J, Weise A, Dowd M, Grant J (2006) Modelling approaches to assess the potential effects of shellfish aquaculture on the marine environment. Canadian Science Advisory Secretariat. Research Document 2006/032.

Commonwealth of Australia (2012) Supporting the Bioregional Plan for the South-west Marine Region: Species Report Cards - Seabirds. Department of Sustainability Environment Water Population and Communities, Canberra, ACT.

Congleton W, Pearce B, Parker M, Beal B (1999) Mariculture siting: a GIS description of intertidal areas. Ecological Modelling 116: 63–75

Cranford P, Ward J, Shumway S (2011) Bivalve filter-feeding: variability and limits of the aquaculture biofilter. In Shumway S. (ed) Shellfish Aquaculture and the Environment. John Wiley and Sons.

Crawford C, Mitchell I, Brown A (1996) Predictive modelling of carrying capacities of oyster (*Crassostrea gigas*) farming areas in Tasmania. Final report to the FRDC. FRDC grant number 92/54.

Crawford C, Macleod K, Mitchell I (2003) Effects of shellfish farming on the benthic environment. Aquaculture 224: 117-140.

Dahlback B, Gunnarsson L (1981) Sedimentation and sulphate reduction under a mussel culture. Marine Biology 63: 269-275.

de Boer WF (2007) Seagrass-sediment interactions, positive feedbacks and critical thresholds for occurrence: A review. Hydrobiologia 591: 5-24.

Government of NSW (2021) NSW Oyster Industry Sustainable Aquaculture Strategy Fourth Edition 2021.

DoF (2016) Fisheries Fact Sheet - Introduced Marine Species. Prepared by the WA Department of Fisheries, Perth Western Australia. Fisheries Fact Sheet No 18, January 2016.

DoH (2016) Marine Biotoxin Monitoring and Management Plan 2016. Western Australian Department of Health, Perth, Western Australia.

DoH (2017) Western Australian Shellfish Quality Assurance Program Operations Manual 2017. Western Australian Department of Health, Perth, Western Australia.

DoW (2013a) WA Oyster Harbour Seagrass 1988. Department of Water, Government of Western Australia. Available online at http://metadata.imas.utas.edu.au/geonetwork/srv/en/metadata.show?uuid=23053B66-472A-41A9-BBD1-85EFCC443065 [Accessed in January 2020].

DoW (2013b) WA Oyster Harbour Seagrass 1996. Department of Water, Government of Western Australia. Available online at http://metadata.imas.utas.edu.au/geonetwork/srv/en/metadata.show?uuid=C17373F8-C646-41CB-A0A5-11A9559251D1 [Accessed in January 2020].

DoW (2013c) WA Oyster Harbour Seagrass 2006. Department of Water, Government of Western Australia. Available online at http://metadata.imas.utas.edu.au/geonetwork/srv/en/metadata.show?uuid=1D34BD8E-74B1-41EB-8F81-398539215527 [Accessed in January 2020].

DPaW (2006) Marine habitats of western Australia database. Available online at http://metadata.imas.utas.edu.au/geonetwork/srv/en/metadata.show?uuid=dfeb72ec-e314-4e6f-9ac9-96c7b1c69aae [Accessed in January 2020].

DPaW (2016) Esperance and Recherche Parks and Reserves: Management Plan 84.

DWER (2020) Water Information Reporting tool. Available online at https://www.water.wa.gov.au/maps-and-data/monitoring/water-information-reporting [Accessed in January 2020].

DWER (2021) Seagrasses in four estuaries in Western Australia's South West 2017-20. Report No. 86. Water Science Technical Series. January 2021.

Ecologia (2007) Albany Iron Ore Project Public Environmental Review – Albany Port Expansion Proposal EPA Assessment No 1594. Prepared for Albany Port Authority by Ecologia Environment, Perth, Western Australia, September 2007.

Ecologia (2009) Albany Iron Ore Project - Albany Port Expansion Proposal Public Environmental Review - Dredge and Land Reclamation EMP. Report: EPA Assessment No 1594, Prepared by Ecologia on behalf of the Albany Port Authority.

EPA (2000) Esperance Port upgrade of marine facilities, incorporating change to existing Environmental Conditions and technical advice in support of Environmental Protection Noise Regulation 17 Approval Notice. EPA Bulletin 999. Western Australian Environmental Protection Authority, Perth Western Australia.

EPA (2016a) Environmental Factor Guideline: Benthic Communities and Habitats. Western Australian Environmental Protection Authority, Perth Western Australia.

EPA (2016b) Technical Guidance: Protection of Benthic Communities and Habitats. Western Australian Environmental Protection Authority, Perth Western Australia.

EPA (2016c) Environmental Factor Guideline, Marine Fauna. Western Australian Environmental Protection Authority, Perth Western Australia

EPA (2016d) Environmental Factor Guideline, Marine Environmental Quality. Western Australian Environmental Protection Authority, Perth Western Australia

EPA (2016e) Technical Guidance Protection of Benthic Communities and Habitats. Western Australian Environmental Protection Authority, Perth Western Australia

EPA (2018) Statement of Environmental Principles, Factors and Objectives. Western Australian Environmental Protection Authority, Perth Western Australia.

EPA (2020) Proposed South Coast Aquaculture Development Zone Letter. Western Australian Environmental Protection Authority, Perth Western Australia.

Ferreira JG, Hawkins AJS, Monteiro P, Moore H, Service M, Pascoe PL, Ramos L, Sequeira A (2008) Integrated assessment of ecosystem-scale carrying capacity in shellfish growing areas. Aquaculture 275:138-151.

GEMS (2007) Grange Resources Albany Port Authority Port Development Oceanographic Studies and Dredging Program Simulated Studies. Report: 376/06, Perth Western Australia.

Gillies CL, McLeod IM, Alleway HK, Cook P, Crawford C, Creighton C, Diggles B, Ford J, Hamer P, Heller-Wagner G, Lebrault E, Le Port A, Russel K, Sheaves M, Warnock B (2018) Australian shellfish ecosystems: Past distribution, current status and future direction. PLOS ONE 13(2): e0190914. https://doi.org/10.1371/journal.pone.0190914 s ,

Goseberg N, Chambers M, Heasman K, Fredriksson D, Fredheim A, Schlurmann T (2017) Technological approaches to longline- and cage-based aquaculture in open ocean environments. In: Buck B., Langan R. (eds) Aquaculture Perspective of Multi-Use Sites in the Open Ocean. Springer, Cham.

Grant J, Hatcher A, Scott D, Pocklington P, Schafer C, Winters G (1995) A multidisciplinary approach to evaluating impacts of shellfish aquaculture on benthic communities. Estuaries 18: 124-144.

Grant J, Cranford P, Hargrave B, Carreau M, Schofield B, Armsworthy S, Burdett-Coutts V, Ibarra D (2005) A model of aquaculture biodeposition for multiple estuaries and field validation at blue mussel (*Mytilus edulis*) culture sites in eastern Canada. Canadian Journal of Fisheries and Aquatic Sciences 62:1271-1285.

Hargrave B, Doucette L, Cranford P, Law B, Milligan T (2008) Influence of mussel aquaculture on sediment organic enrichment in a nutrient-rich coastal embayment. Marine Ecology Progress Series 365: 137-149.

Harper EM (1996) Attachment of mature oysters (*Saccostrea cucullata*) to natural substrata. Marine Biology 127:449–553.

Hayakawa Y, Kobayashi M, Izawa M (2001) Sedimentation flux from mariculture of oyster (*Crassostrea gigas*) in Ofunato estuary, Japan. ICES Journal of Marine Science 58: 435-444.

Hillman K, Lukatelich R, Bastyan G, McComb A (1991) Water quality and seagrass biomass, productivity and epiphyte load in Princess Royal Harbour, Oyster Harbour and King George Sound. Albany Harbors Environmental Study (1988-1989), Environmental Protection Authority. Perth, Western Australia.

Joyce A, Rubio A and Winberg P (2010) Environmental and Socio-Economic Considerations for Aquaculture in Jervis Bay, NSW. Canberra: Fisheries Research and Development Corporation.

Kamiyama T, Nagai S, Suzuki T, Miyamura K (2010) Effect of temperature on production of okadaic acid, dinophysistoxin-1, and pectenotoxin-2 by *Dinophysis acuminata* in culture experiments. Aquatic Microbial Ecology 60: 193-202.

Kang CK, Choy EJ, Hur YB, Myeong JI (2009) Isotopic evidence of particle size-dependent food partitioning in cocultured sea squirt *Halocynthia roretzi* and Pacific oyster *Crassostrea gigas*. Aquatic Biology 6:289-302.

Kilminster K, Hovey R, Waycott M, Kendrick GA (2018) Seagrass of Southern and South-Western Australia. In Larkum A, Kendrick GA, Ralph P (eds), Seagrasses of Australia: Structure, Ecology and Conservation p61–89. Netherlands: Springer.

Kirkman H (1997) Seagrasses of Australia. CSIRO, Canberra ACT.

Kohan A, Nasrolahi A, Aeinjamshid K, Kiabi BH (2019) Potential use of two fouling species (*Amphibalanus Amphitrite* and *Saccostrea cucculata*) to prevent shrimp farm-induced coastal eutrophication. Ocean and Coastal Management 173:10-16.

Liao Y, Shou L, Jiang Z, Tang Y, Du P, Zeng J, Chen Q, Yan X, Chen J (2019) Effects of fish cage culture and suspended oyster culture on macrobenthic communities in Xiangshan Bay, a semi-enclosed subtropical bay in eastern China. Marine Pollution Bulletin 142: 475-483.

McClatchie S, Middleton J, Pattiaratchi (2006) The South-West Marine Region: Ecosystems and Key Species Groups. Department of the Environment and Water Resources, Commonwealth of Australia, Canberra, ACT.

McKindsey CW, Thetmeyer H, Landry T, Silvert W (2006) Review of recent carrying capacity models for bivalve culture and recommendations for research and management. Aquaculture 261:451-462.

Mills DA, Brady KN (1985) Wind driven circulation in Princess Royal Harbour and Oyster Harbour, Results from a Numerical Model. Prepared for Department of Conservation and Environment.

Mills DA, D'Adamo N (1993) Water circulation and flushing characteristics of Princess Royal Harbour, Albany. Environmental Protection Authority, Perth.

Mitchell I (2006) In situ bio-deposition rates of Pacific Oysters (*Crassostrea gigas*) on a marine farm in Southern Tasmania (Australia). Aquaculture 257: 194-203.

MScience (2019) Habitat Map Validation Survey, MScience report number MSA267R03. Prepared for the Department of Primary Industries and Regional Development.

Numaguchi K (1994) Studies on the feeding ecology and food environment of the Japanese pearl oyster *Pinctada fucata martensii*.

Oceanica (2006) Albany Waterfront Project - Protected Harbour Development Princess Royal Harbour Benthic Habitat Mapping Data Report. Report No. 382/2. Prepared for LandCorp by Oceanica, September 2006.

O'Connor WA, Lawler NF, Heasman MP (2003) Trial farming the Akoya pearl oyster, *Pinctada imbricata*, in Port Stephens, NSW. NSW Fisheries Final Report Series No. 42.

O'Connor WA, Dove MC (2009) The changing face of oyster aquaculture in New South Wales, Australia. Journal of Shellfish Research 28(4):803–811.

OGA (2016) Aquaculture Management and Environmental Monitoring Plan. Prepared by Ocean Grown Abalone Pty Ltd.

O'Neil JM, Davis TW, Burford MA, Gobler CJ (2012) The rise of harmful cyanobacteria blooms: the potential roles of eutrophication and climate change. Harmful Algae 14: 313-344.

Pascoe PL, Parry HE, Hawkins AJS (2009) Observations on the measurement and interpretation of clearance rate variations in suspension-feeding bivalve shellfish. Aquatic Biology 6:181-190.

Phlips EJ, Badylak S, NG Nelson, Havens KE (2020) Hurricanes, El Niño and harmful algal blooms in two sub-tropical Florida estuaries: direct and indirect impacts. Scientific Reports 10.

Serrano O (2016) Seagrass sediments reveal the long-term deterioration of an estuarine ecosystem. Global change biology (1354-1013), 22 (4), p. 1523-1531.

SKM (2007) Sampling and Analysis Plan Report, Albany Iron Ore Project, Albany Port Expansion Proposal. Perth Western Australia.

South Coast Natural Resource Management (2019) Community Consultation Report. Prepared for the Department of Primary Industries and Regional Development.

Strategen (2008) Albany Protected Harbour Development Environmental Protection Statement. Prepared for Landcorp, Perth WA.

The Nature Conservancy (2020) Putting oyster reefs back into Oyster Harbour. Available online at https://www.natureaustralia.org.au/what-we-do/our-priorities/oceans/ocean-stories/putting-the-oyster-reefs-back-into-oyster-harbour/ [Accessed 20 April 2020].

Thomson C (2018) Regional Estuaries Initiative, Estuary Condition Report: Oyster Harbour 2016/17. Department of Water and Environmental Regulation, Western Australia.

Wells FE (ed) (1988) The Marine Flora and Fauna of Albany, Western Australia: Proceedings of the Third International Marine Biological Workshop. Australian Marine Science Association, Western Australian Museum, Perth, WA.

Wells FE, McDonald JI, Huisman JM (2009) Introduced Marine Species in Western Australia. Department of Fisheries, Report No 57, Perth, Western Australia.

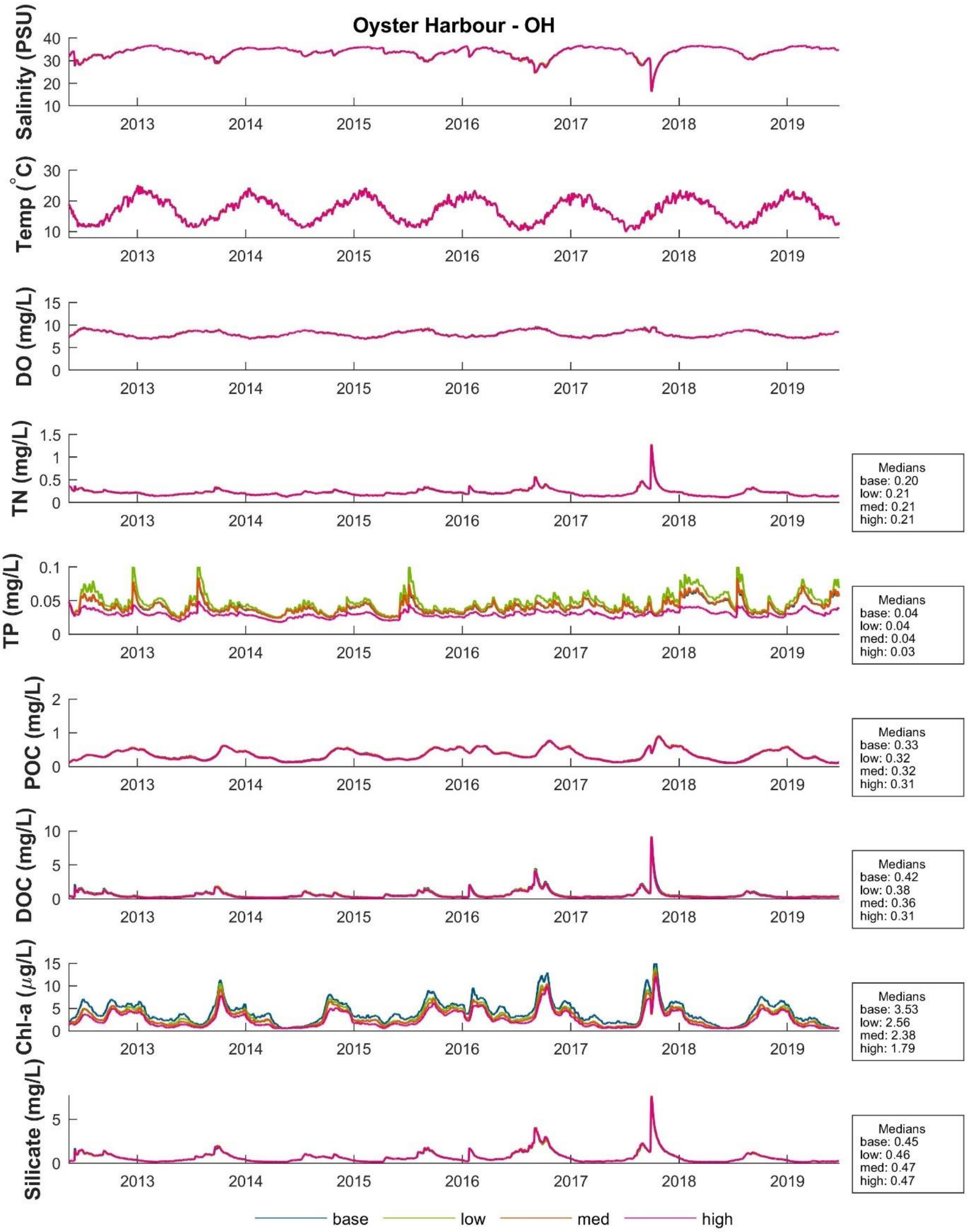
Wetz MS, Yoskowitz DW (2013) An 'extreme' future for estuaries? Effects of extreme climatic events on estuarine water quality and ecology. Marine Pollution Bulletin 69: 7-18.

Zhou Y, Zhang S, Liu Y, Yang H (2014) Biologically induced deposition of fine suspended particles by filter-feeding bivalves in land-based industrial marine aquaculture wastewater. PLoS ONE 9(9): e107798.

Appendix A Carrying Capacity Modelling Timeseries

A.1 Timeseries - Interconnected system

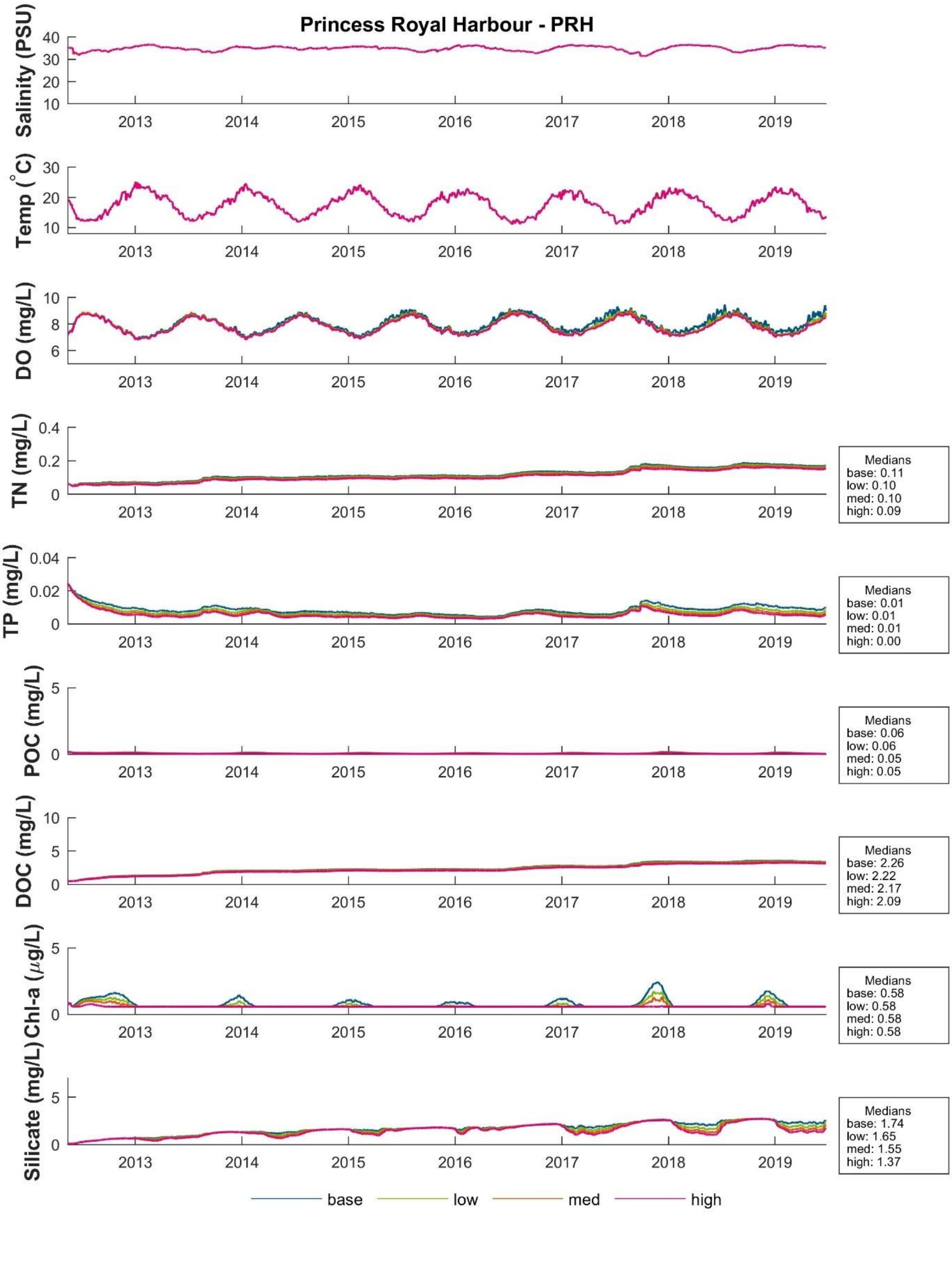
Figures A1 to A9 represent water quality indicators for Oyster Harbour, Princess Royal Harbour and King George Sound under the carrying capacity modelling scenarios detailed in Table 5.9, Table 5.10 and Table 5.11 whereby shellfish were present in each area. Values are given for surface waters.



Notes:

1. Base represents the water quality with current baseline biomasses; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 406, 456 and 506 tonnes (1218, 1368 and 1518 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

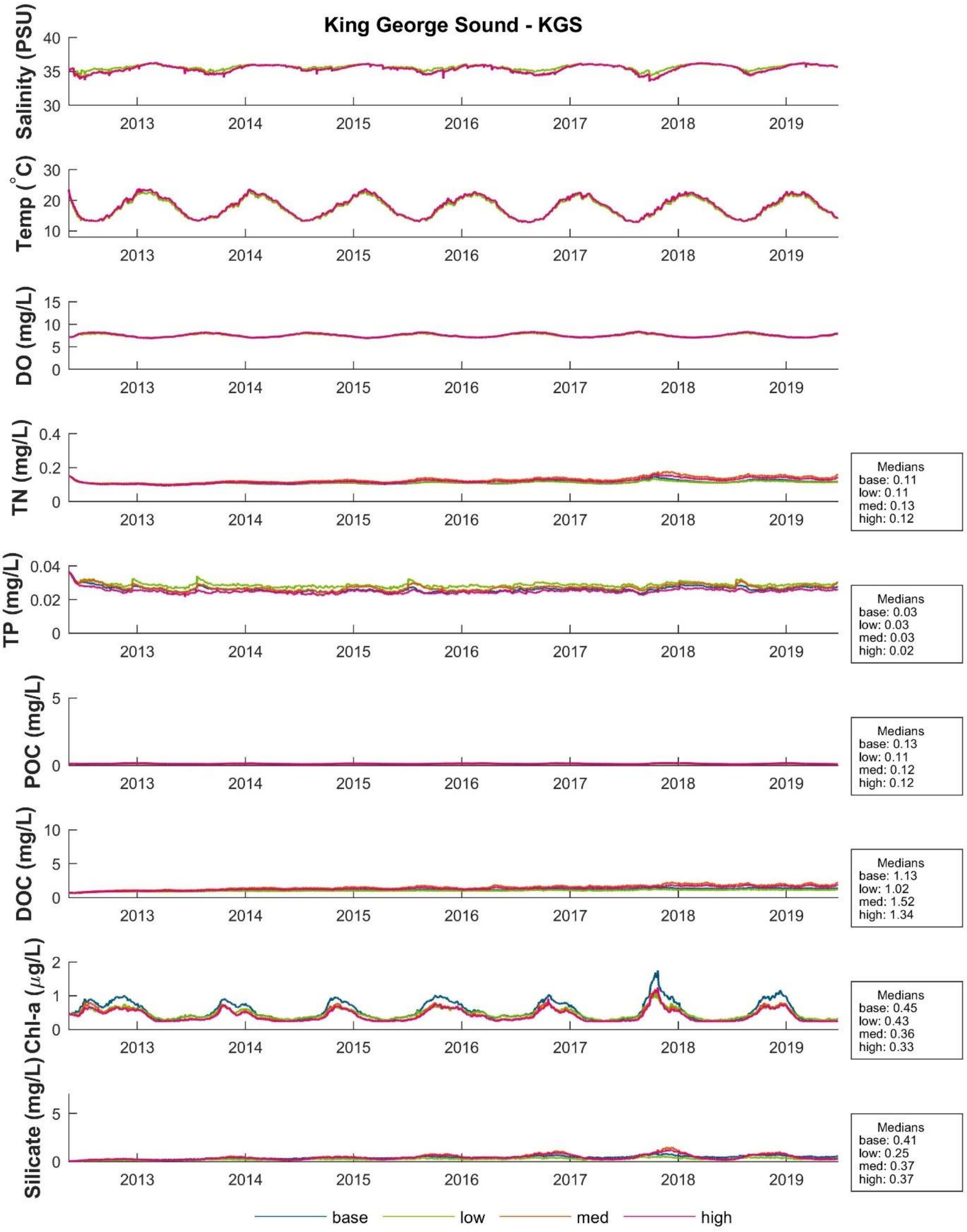
Figure A.1 Water quality indicator results for surface waters of Oyster Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 1.3 L/hr



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 14, 20 and 27 tonnes (40, 60 and 80 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

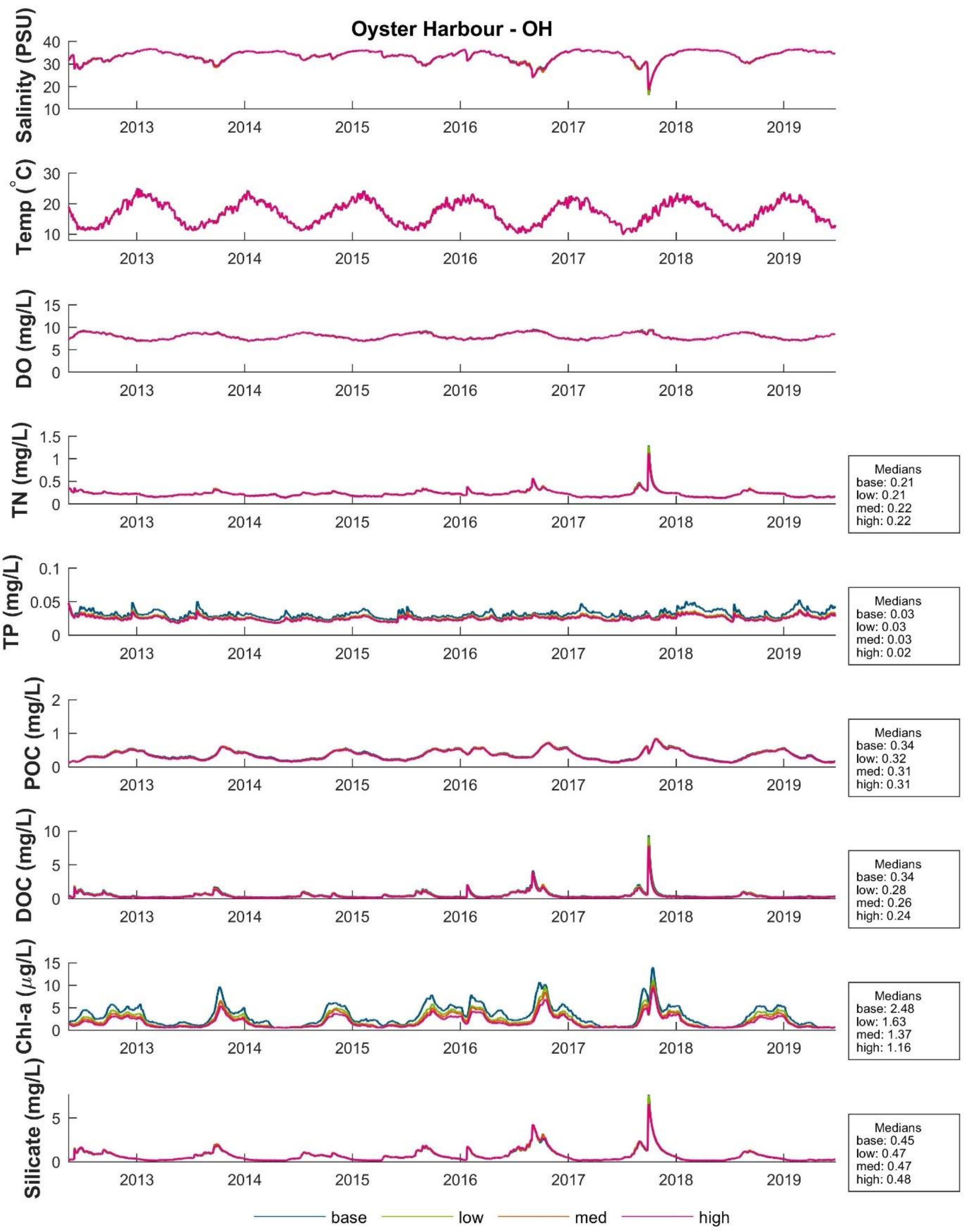
Figure A.2 Water quality indicator results for surface waters of Princess Royal Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 1.3 L/hr



Notes:

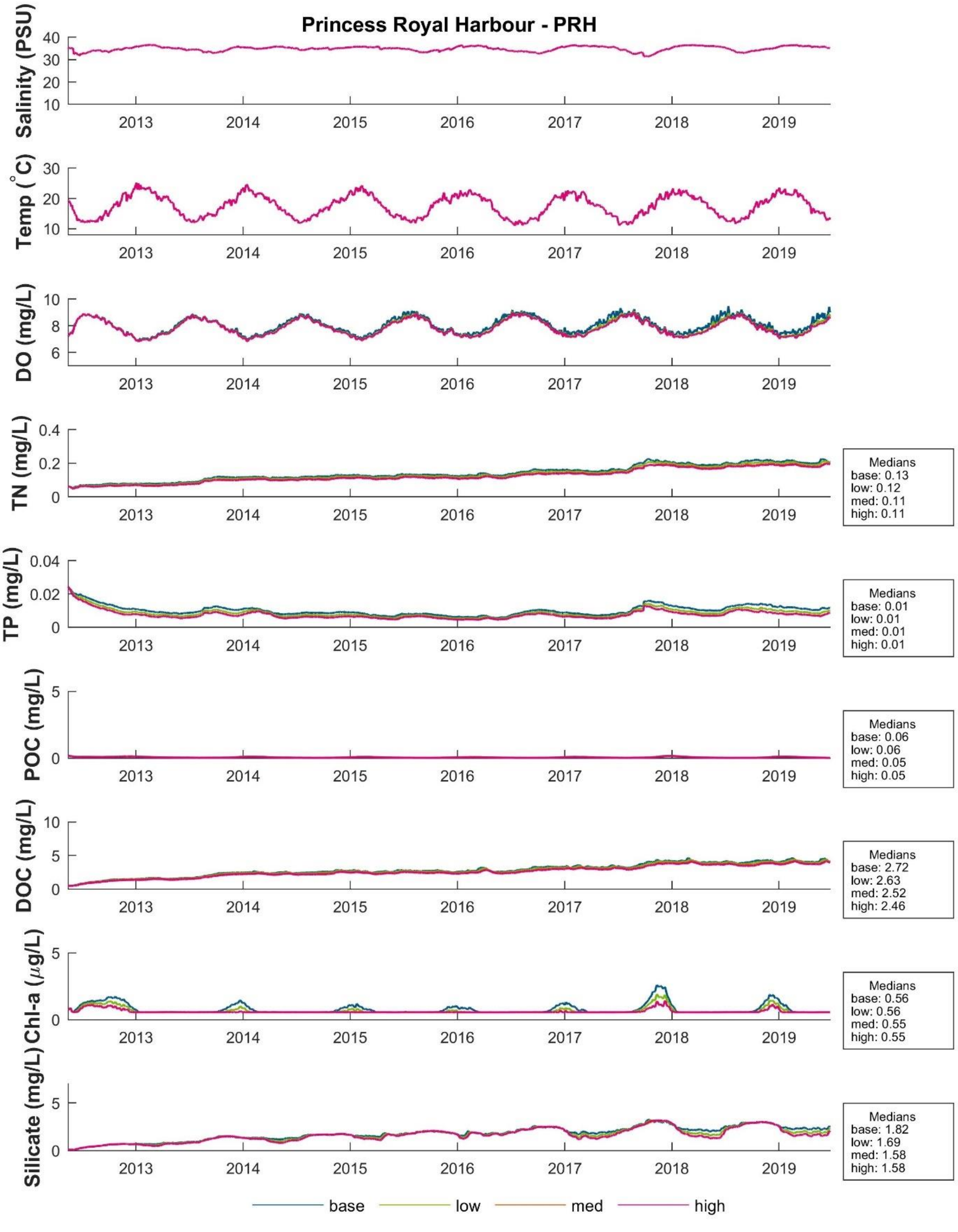
1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production/standing biomass of *P. fucata* and *M. galloprovincialis* equates to 804, 958 and 1111 tonnes respectively.
2. Median results for each scenario are given in boxes on the right.

Figure A.3 Water quality indicator results for surface waters of King George Sound with increasing biomasses of *Pinctada imbricata fucata* and *Mytilus galloprovincialis* using clearance rates of 5 and 1.3 L/hr respectively



Notes:
1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 406, 456 and 506 tonnes (1218, 1368 and 1518 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

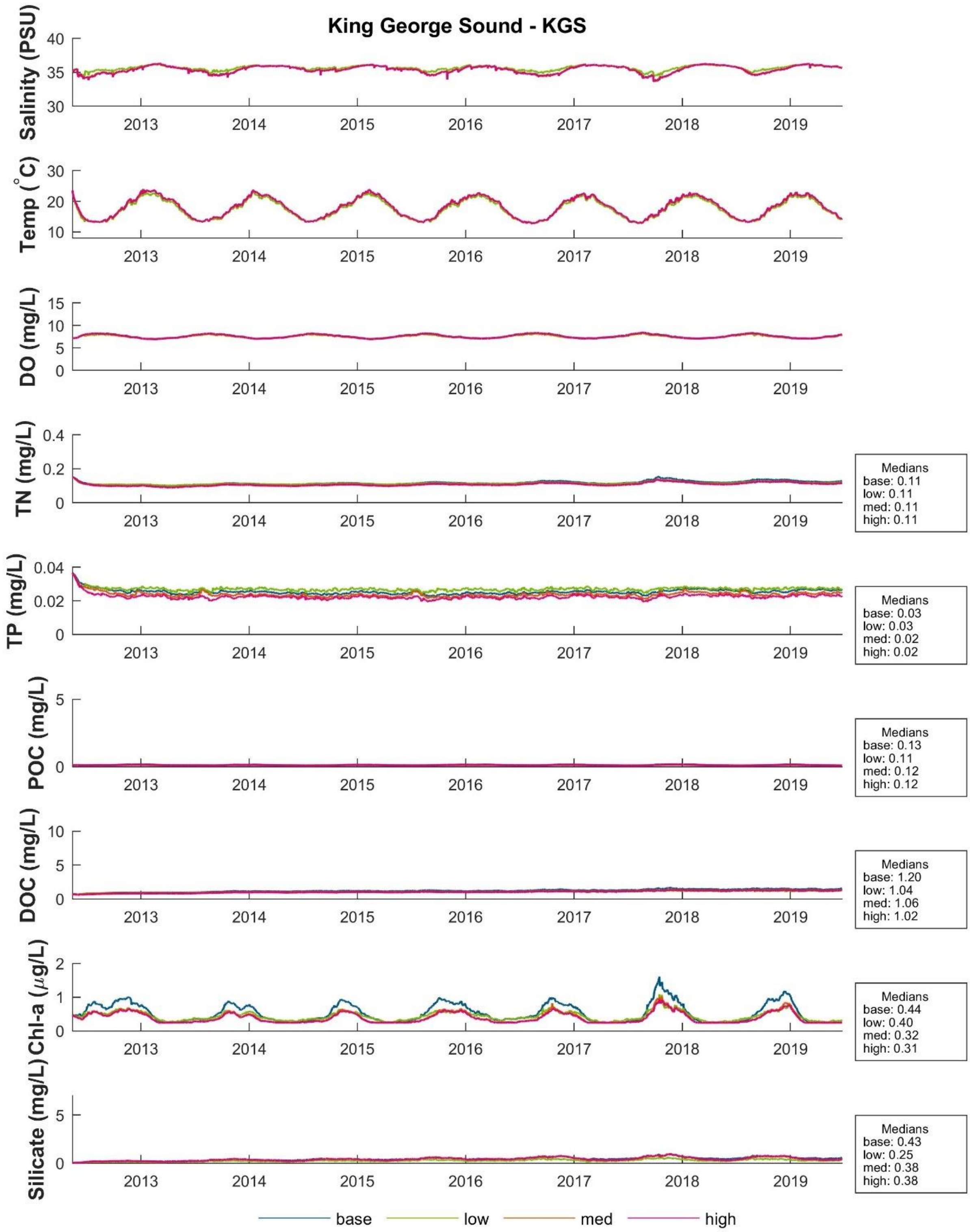
Figure A.4 Water quality indicator results for surface waters of Oyster Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 2.2 L/hr



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 14, 20 and 27 tonnes (40, 60 and 80 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

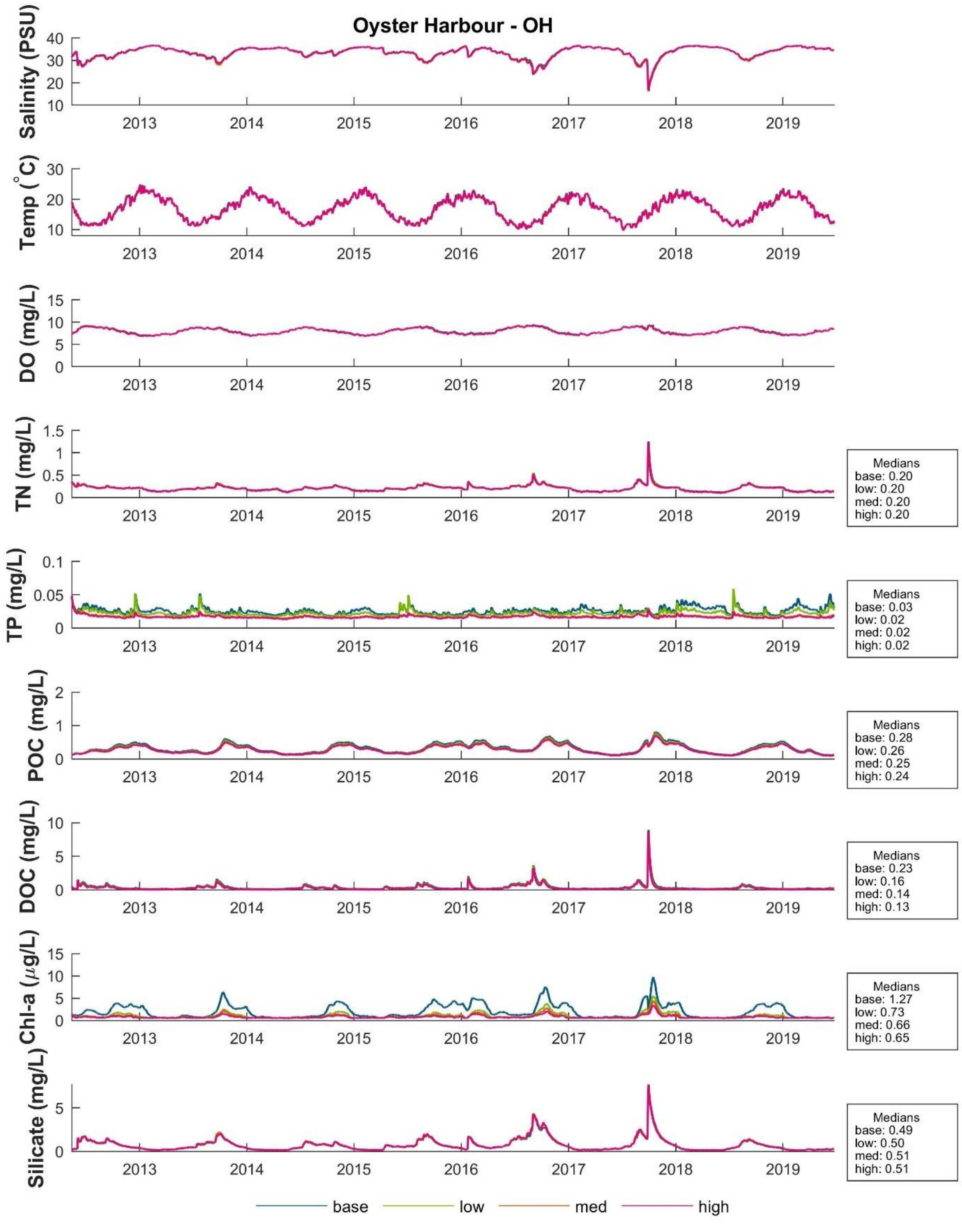
Figure A.5 Water quality indicator results for surface waters of Princess Royal Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 2.2 L/hr



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production/standing biomass of *P. fucata* and *M. galloprovincialis* equates to 804, 958 and 1111 tonnes respectively.
2. Median results for each scenario are given in boxes on the right.

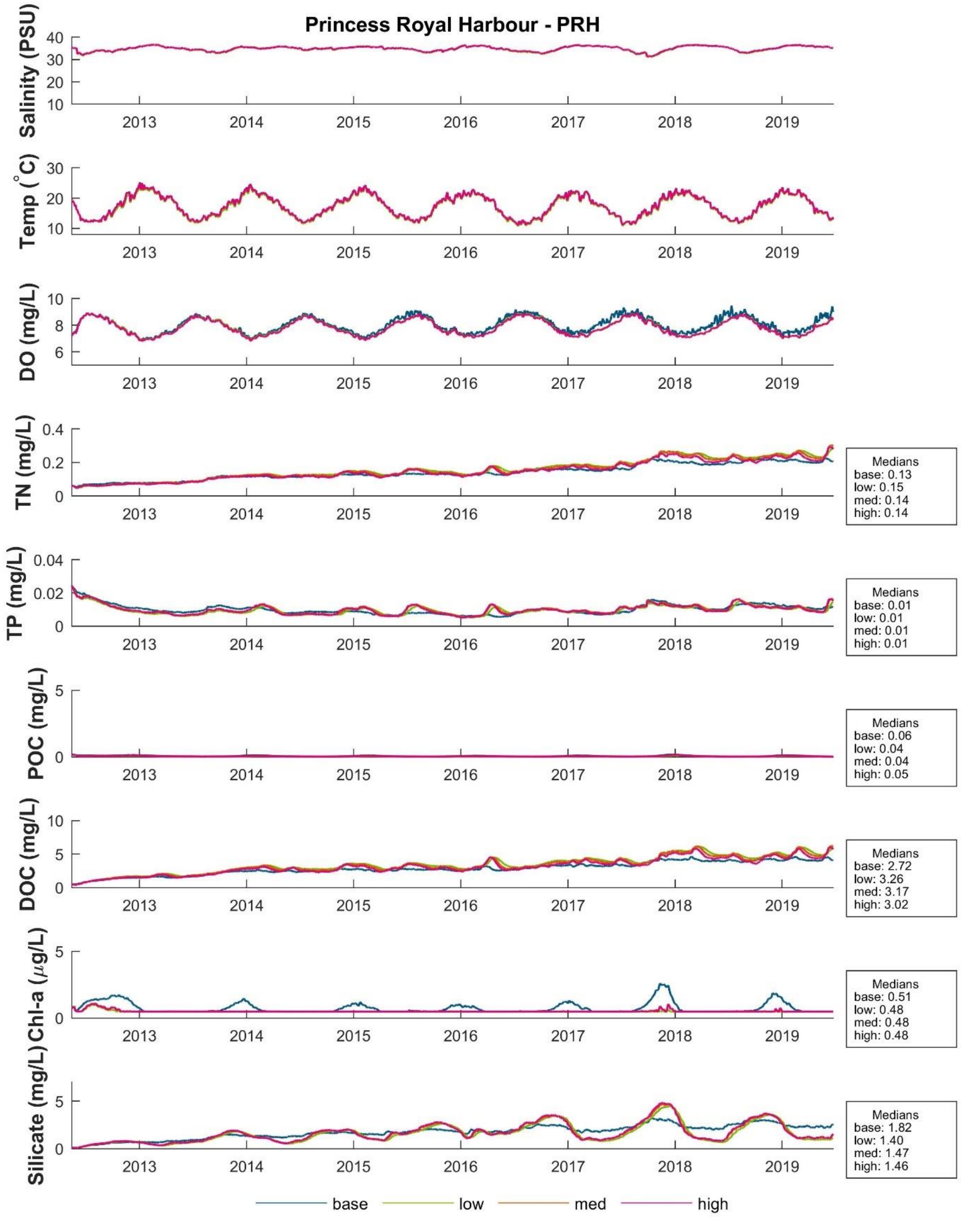
Figure A.6 Water quality indicator results for surface waters of King George Sound with increasing biomasses of *Pinctada imbricata fucata* and *Mytilus galloprovincialis* using clearance rates of 15 and 2 L/hr respectively



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 406, 456 and 506 tonnes (1218, 1368 and 1518 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

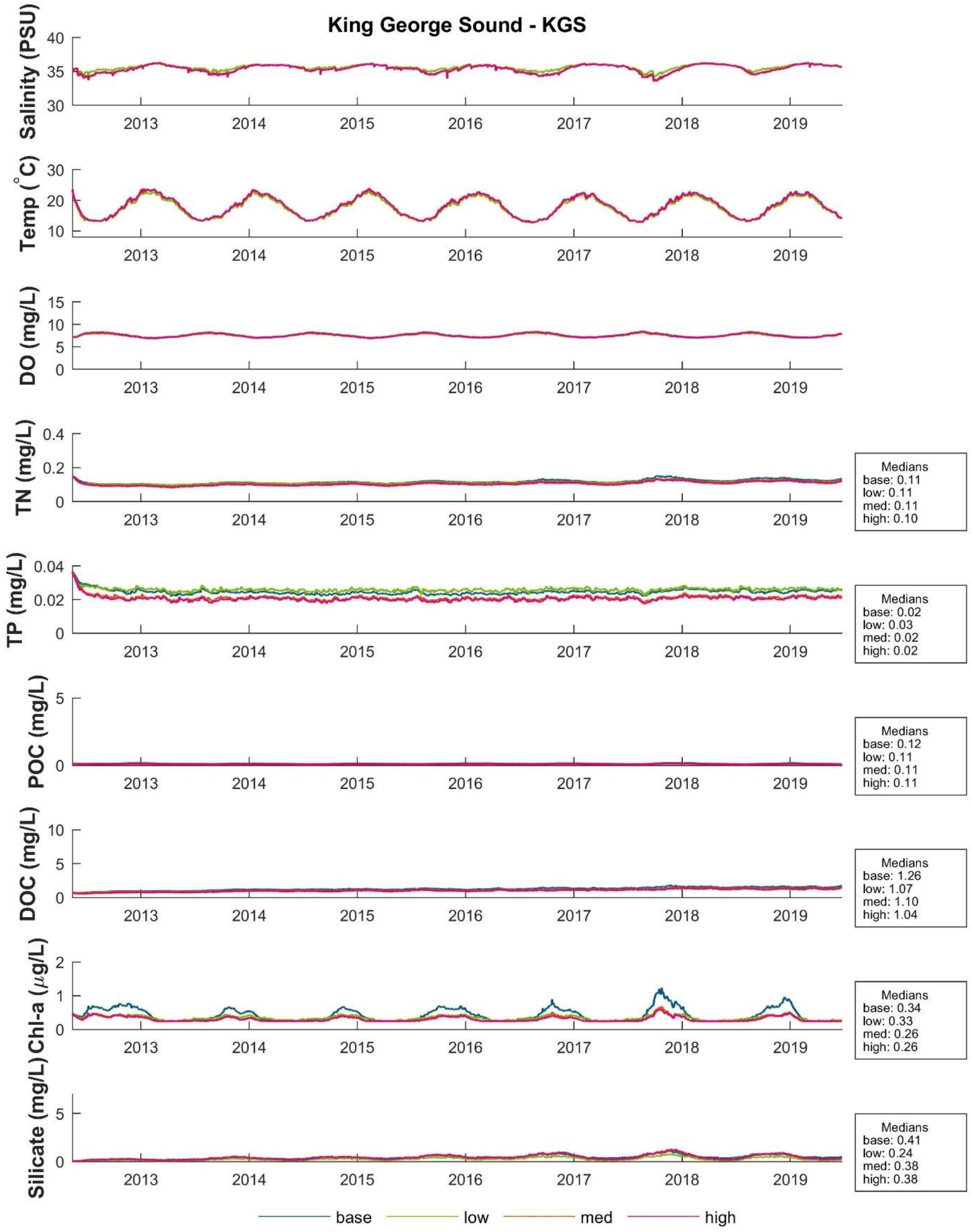
Figure A.7 Water quality indicator results for surface waters of Oyster Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 4 L/hr



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 14, 20 and 27 tonnes (40, 60 and 80 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

Figure A.8 Water quality indicator results for surface waters of Princess Royal Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 4 L/hr



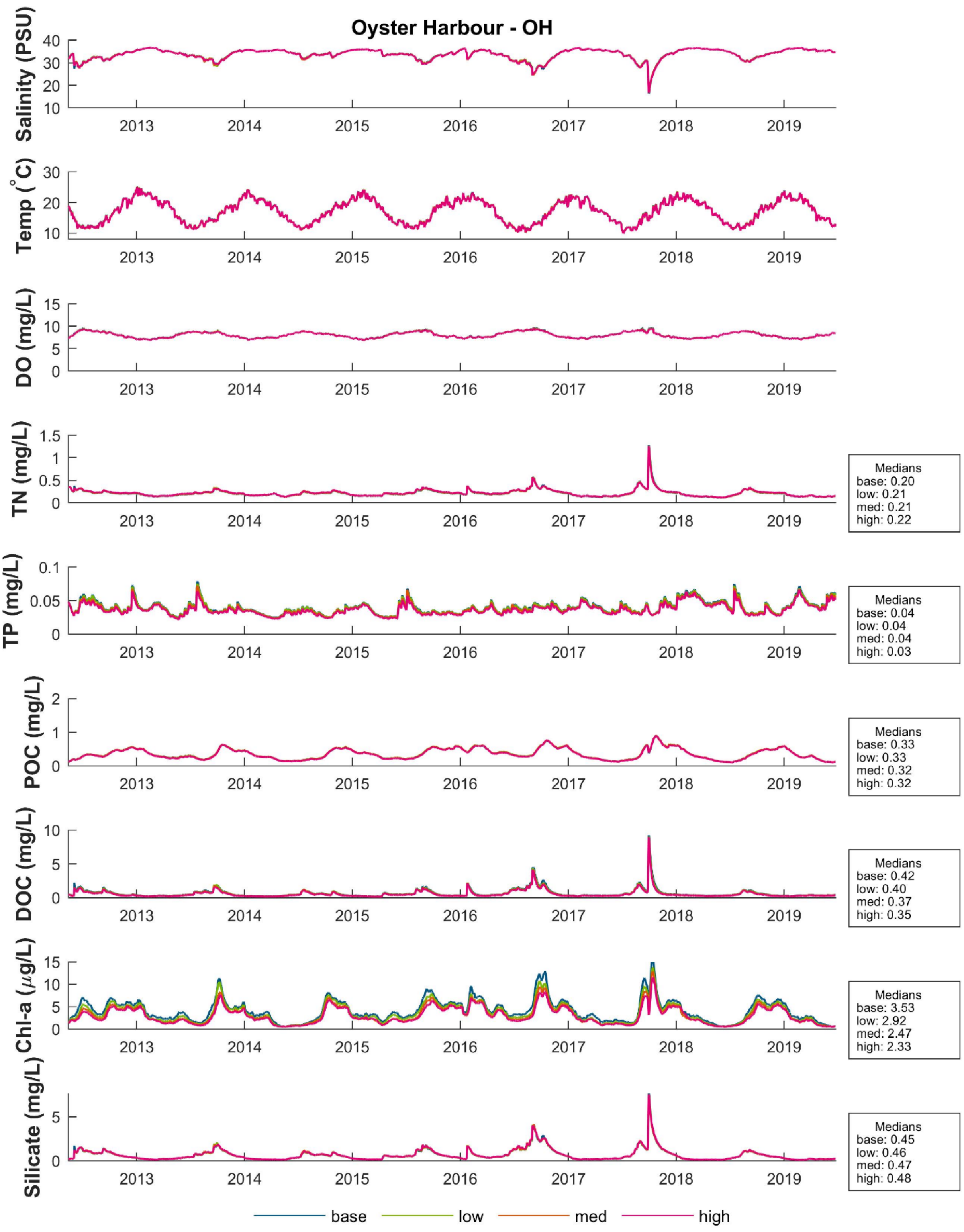
Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production/standing biomass of *P. fucata* and *M. galloprovincialis* equates to 804, 958 and 1111 tonnes respectively.
2. Median results for each scenario are given in boxes on the right.

Figure A.9 Water quality indicator results for surface waters of King George Sound with increasing biomasses of *Pinctada imbricata fucata* and *Mytilus galloprovincialis* using clearance rates of 30 and 4 L/hr respectively

A.2 Timeseries - Independent systems

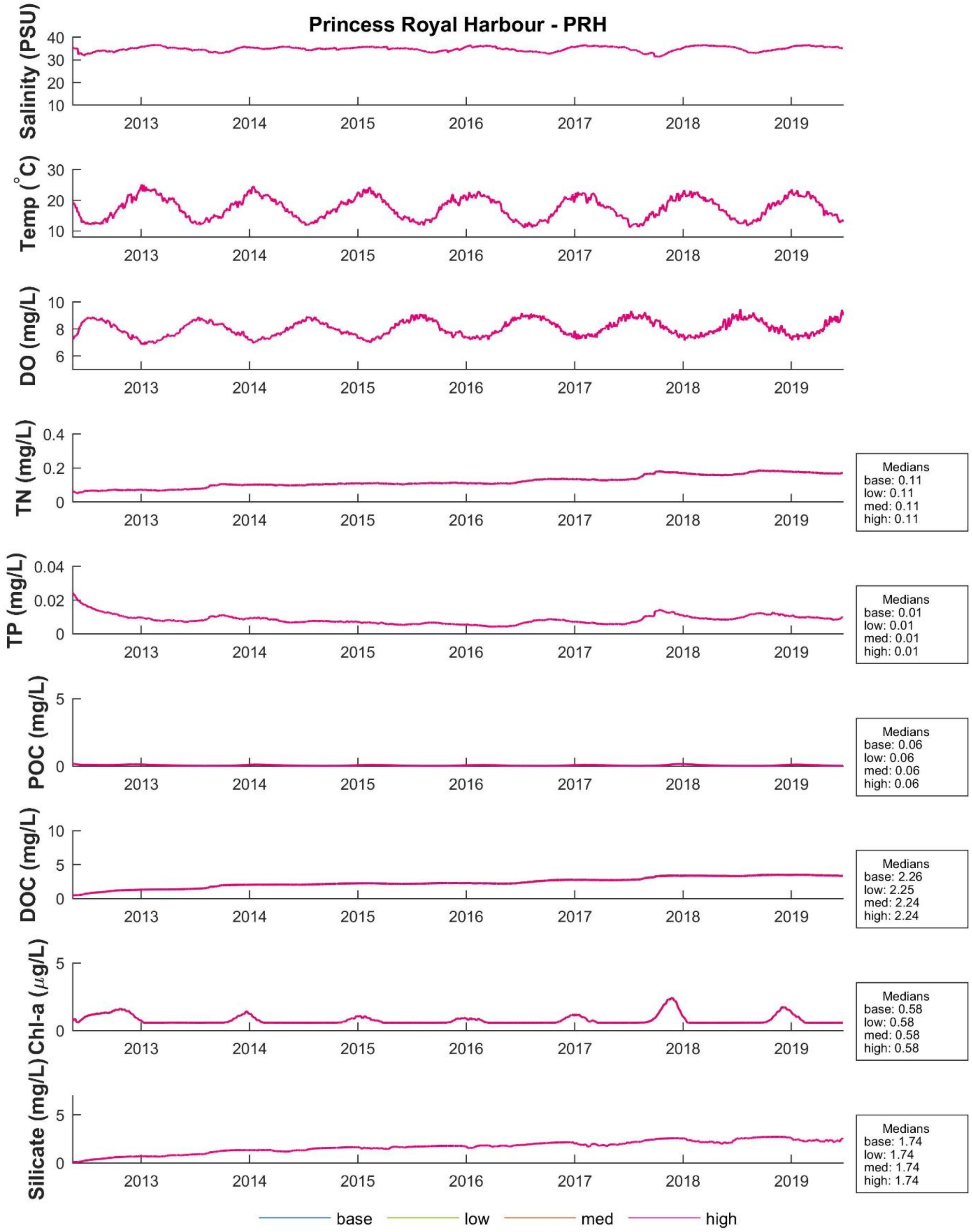
Figures A10 to A19 represent water quality indicators for Oyster Harbour, Princess Royal Harbour and King George Sound under the carrying capacity modelling scenarios detailed in Table 5.9, Table 5.10 and Table 5.11 whereby shellfish were present only in the area being directly modelled. Values are given for surface waters.



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 406, 456 and 506 tonnes (1218, 1368 and 1518 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

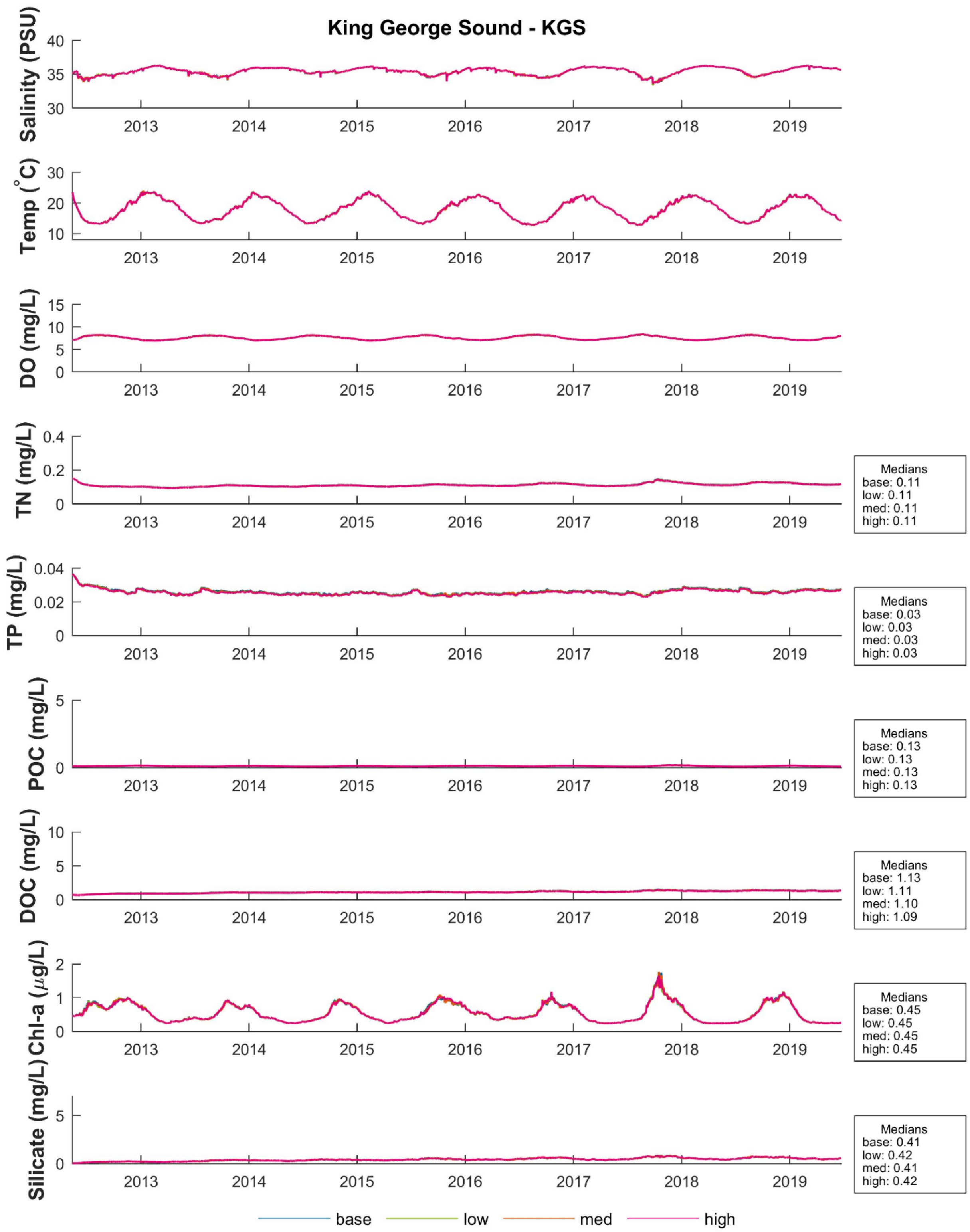
Figure A.10 Water quality indicator results for surface waters of Oyster Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 1.3 L/hr and only current baseline biomasses of shellfish present in other areas



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 14, 20 and 27 tonnes (40, 60 and 80 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

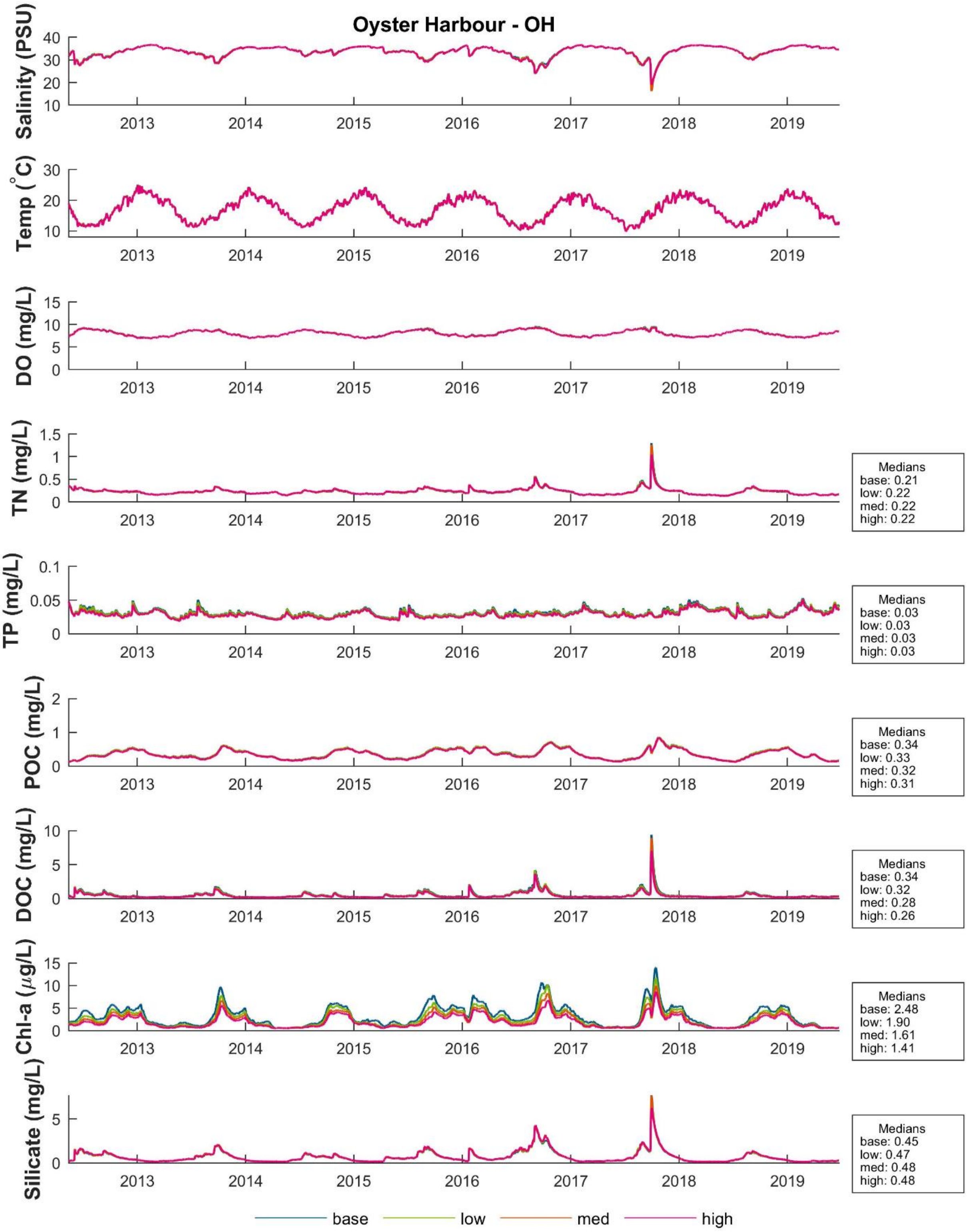
Figure A.11 Water quality indicator results for surface waters of Princess Royal Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 1.3 L/hr and only current baseline biomasses of shellfish present in other areas.



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production/standing biomass of *P. fucata* and *M. galloprovincialis* equates to 804, 958 and 1111 tonnes respectively.
2. Median results for each scenario are given in boxes on the right.

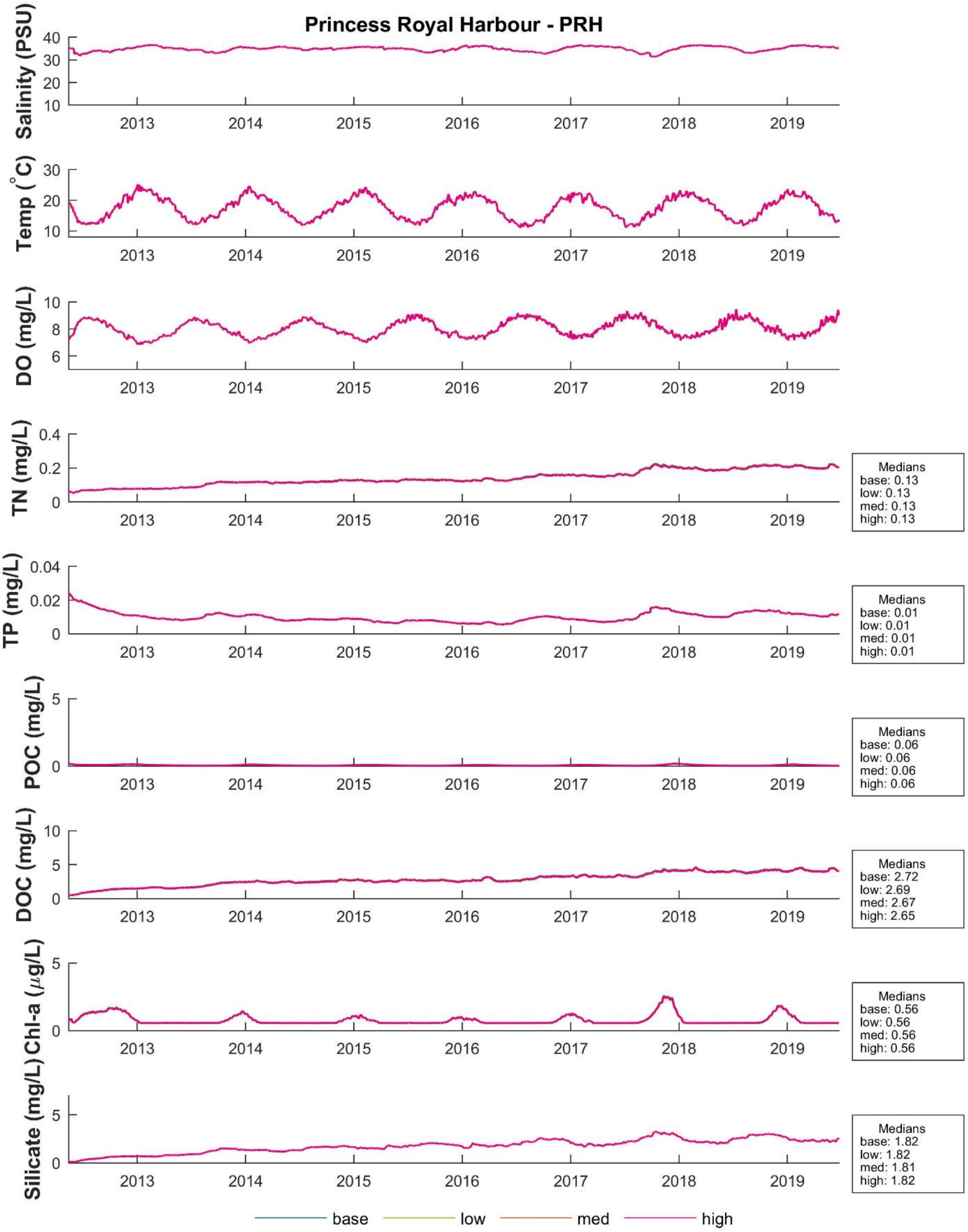
Figure A.12 Water quality indicator results for surface waters of King George Sound with increasing biomasses of *Pinctada imbricata fucata* and *Mytilus galloprovincialis* using clearance rates of 5 and 1.3 L/hr respectively and only current baseline biomasses of shellfish present in other areas.



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 406, 456 and 506 tonnes (1218, 1368 and 1518 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

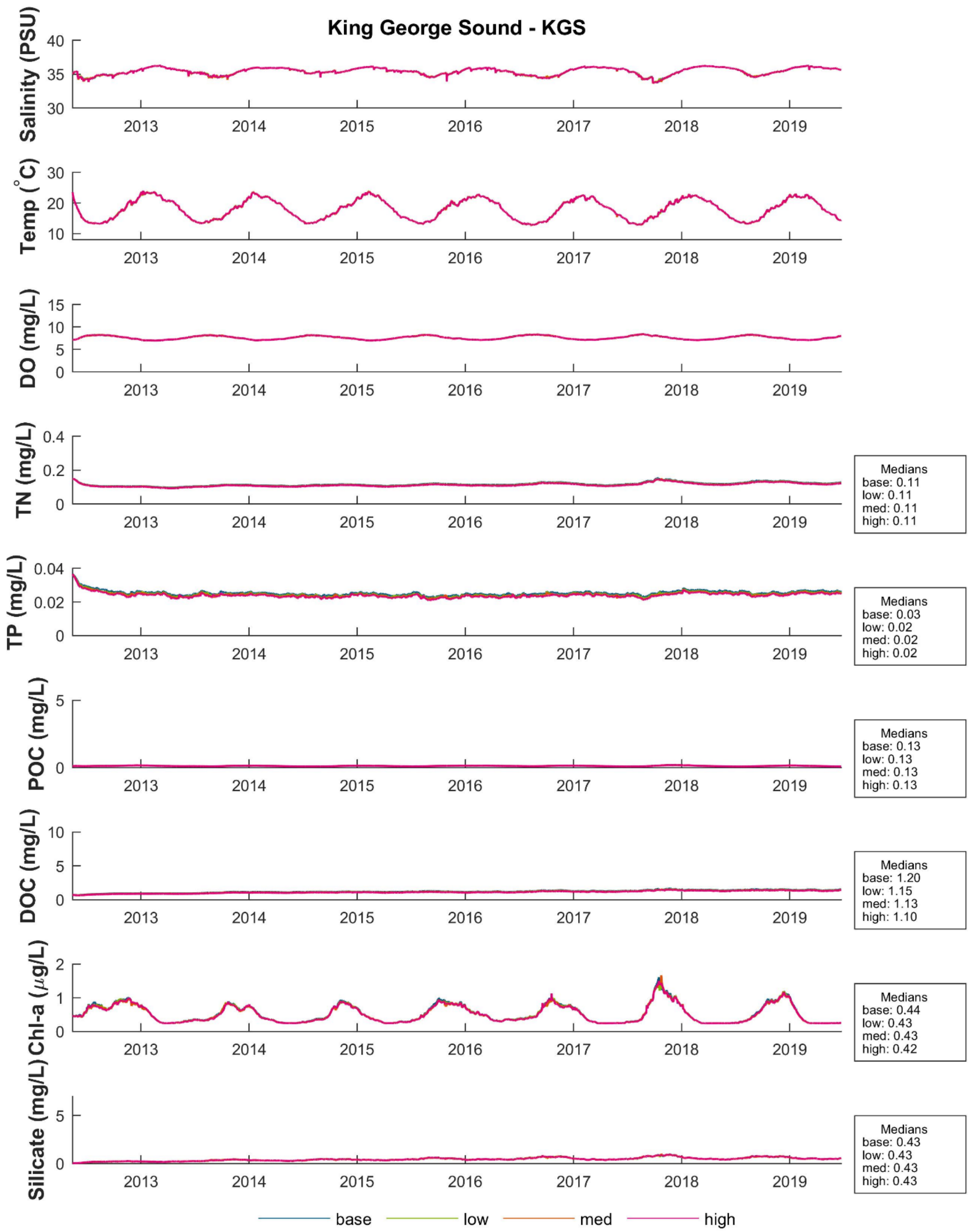
Figure A.13 Water quality indicator results for surface waters of Oyster Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 2.2 L/hr and only current baseline biomasses of shellfish present in other two areas.



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 14, 20 and 27 tonnes (40, 60 and 80 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

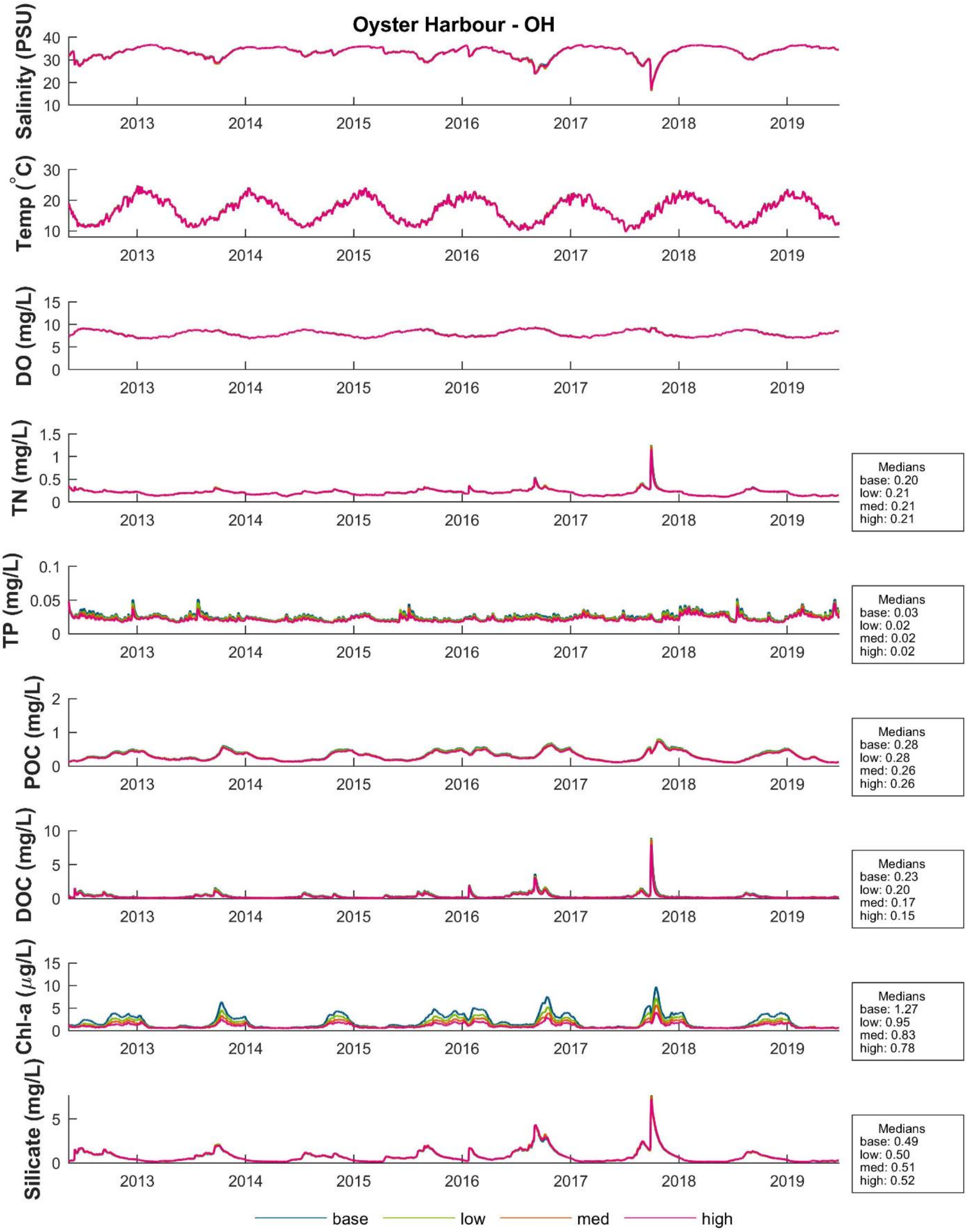
Figure A.14 Water quality indicator results for surface waters of Princess Royal Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 2.2 L/hr and only current baseline biomasses of shellfish present in other areas.



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production/standing biomass of *P. fucata* and *M. galloprovincialis* equates to 804, 958 and 1111 tonnes respectively.
2. Median results for each scenario are given in boxes on the right.

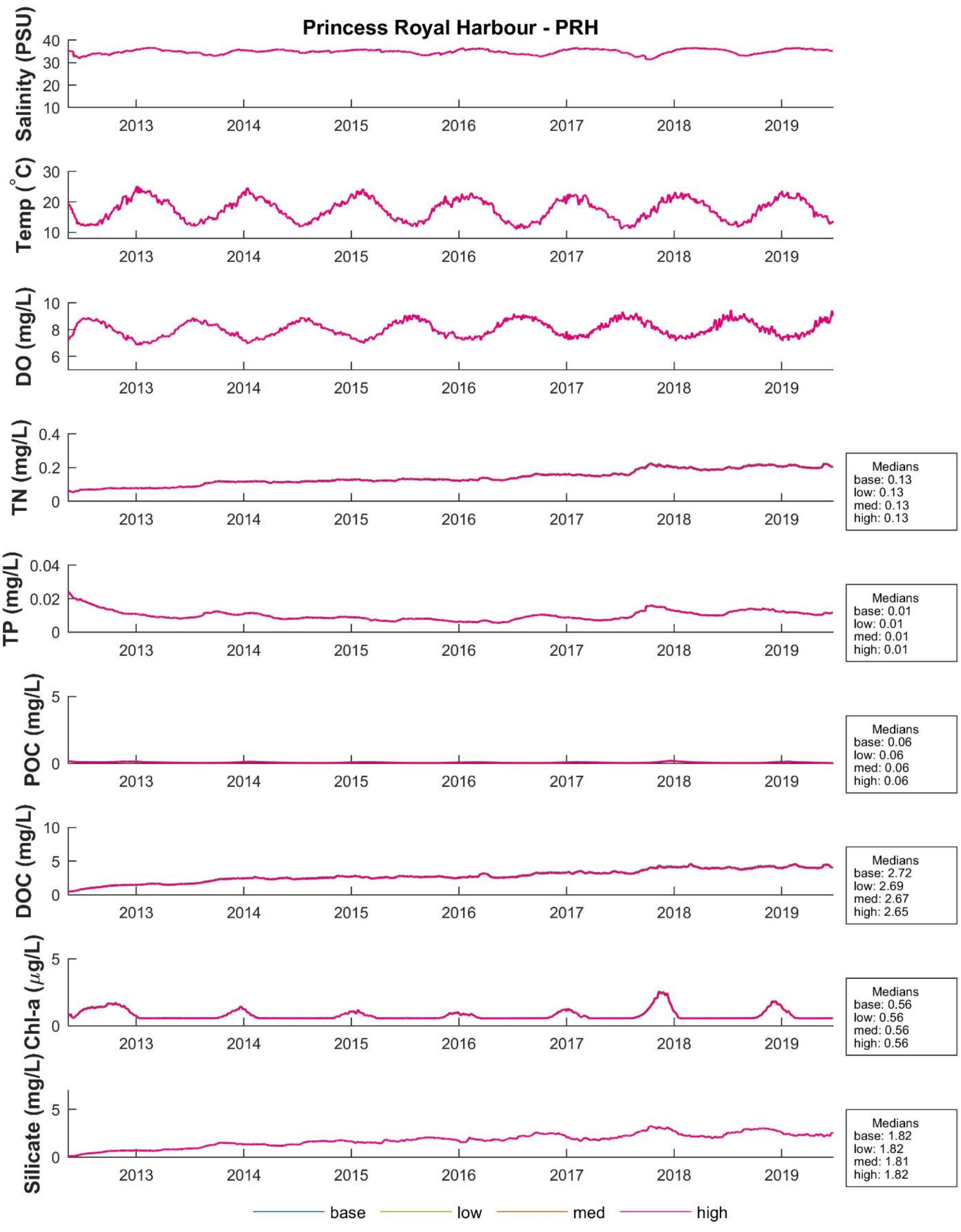
Figure A.15 Water quality indicator results for surface waters of King George Sound with increasing biomasses of *Pinctada imbricata fucata* and *Mytilus galloprovincialis* using clearance rates of 15 and 2 L/hr respectively and only current baseline biomasses of shellfish present in other areas.



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 406, 456 and 506 tonnes (1218, 1368 and 1518 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

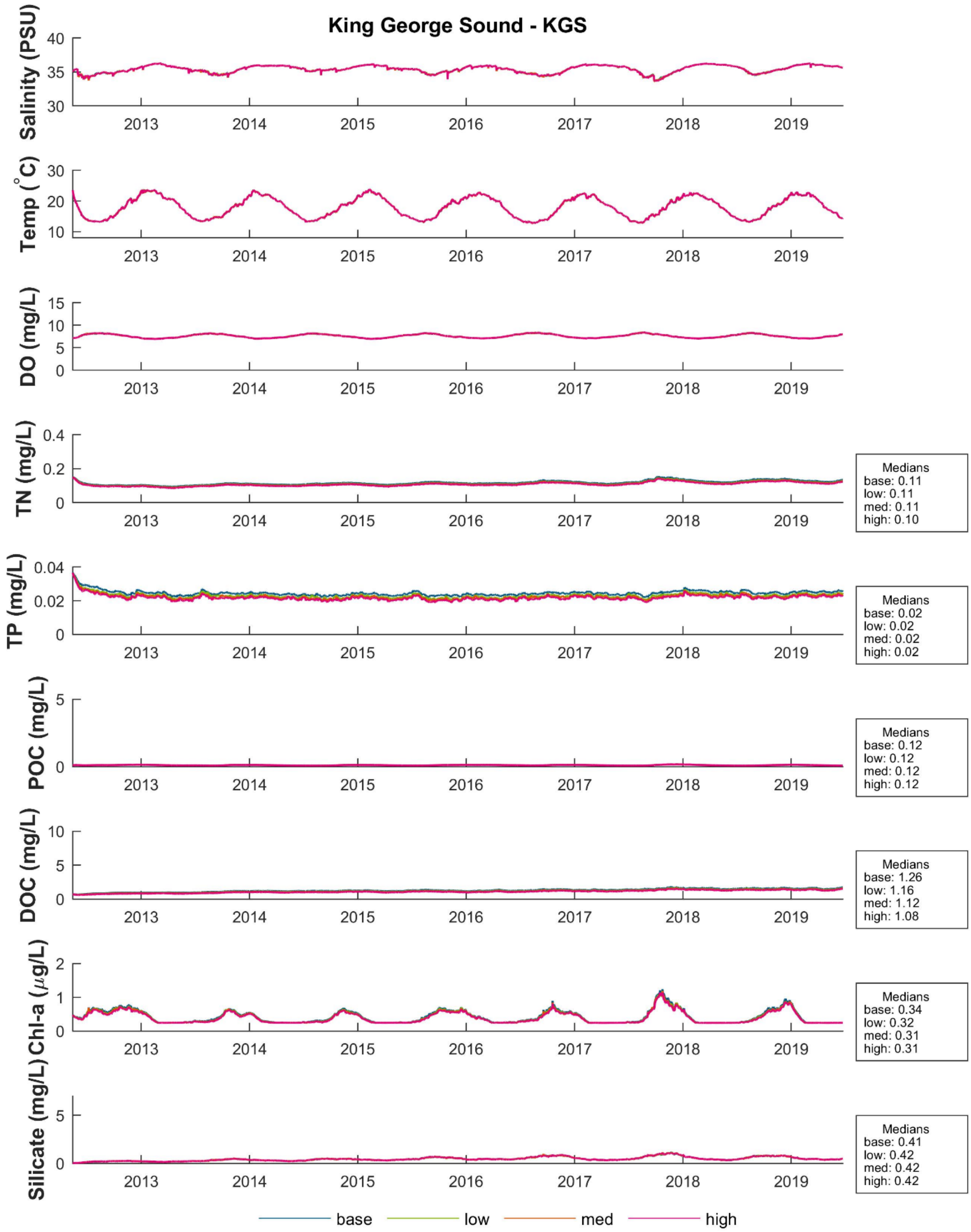
Figure A.16 Water quality indicator results for surface waters of Oyster Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 4 L/hr and only current baseline biomasses of shellfish present in other two areas.



Notes:

1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production of *S. glomerata* equates to 14, 20 and 27 tonnes (40, 60 and 80 tonnes standing biomass) respectively.
2. Median results for each scenario are given in boxes on the right.

Figure A.17 Water quality indicator results for surface waters of Princess Royal Harbour with increasing biomasses of *Saccostrea glomerata* using a clearance rate of 4 L/hr and only current baseline biomasses of shellfish present in other areas.



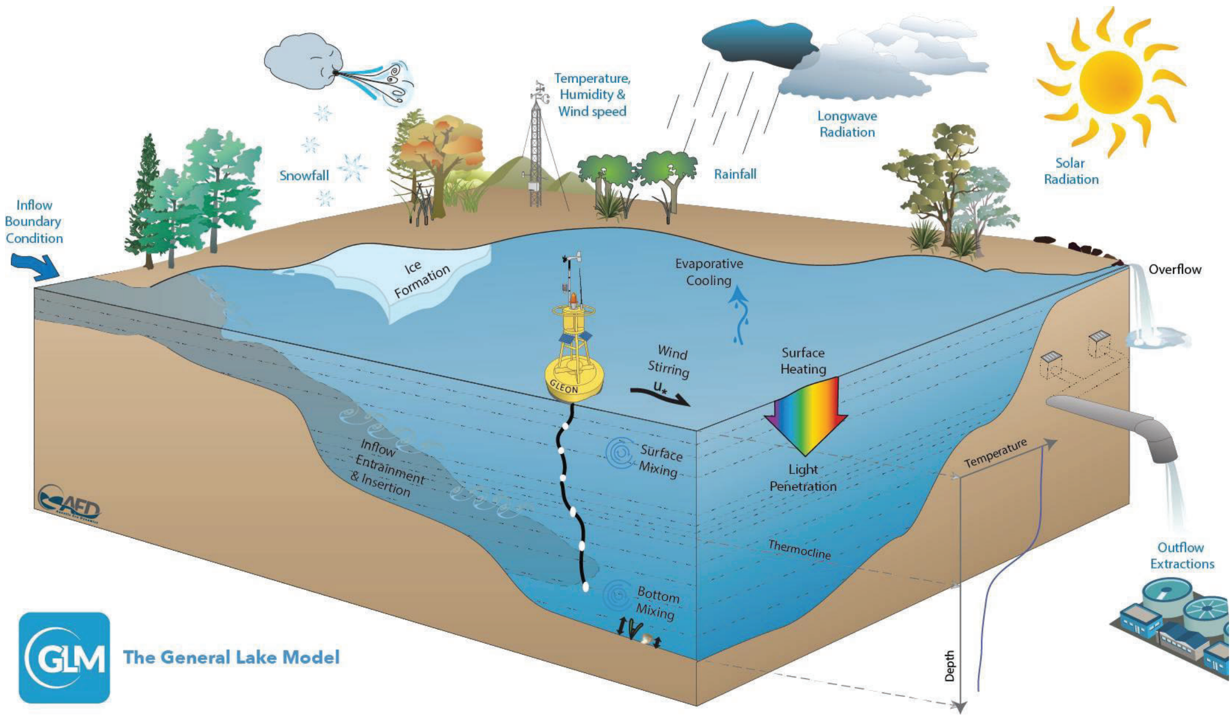
Notes:
1. Base represents the water quality before the addition of shellfish; low, med and high represent the scenarios where annual production/standing biomass of *P. fucata* and *M. galloprovincialis* equates to 804, 958 and 1111 tonnes respectively.
2. Median results for each scenario are given in boxes on the right.

Figure A.18 Water quality indicator results for surface waters of King George Sound with increasing biomasses of *Pinctada imbricata fucata* and *Mytilus galloprovincialis* using clearance rates of 30 and 4 L/hr respectively and only current baseline biomasses of shellfish present in other areas.

Appendix B Modelling Tools

GLM (General Lake Model)

GLM (General Lake Model) is a 1D hydrodynamic model that solves water, thermal and mass balances. GLM computes vertical profiles of temperature, salinity and density by accounting for the effect of inflows/outflows, as well as meteorological influences on surface heat fluxes and mixing (and therefore stratification) of the lake. The model is one-dimensional; hence it assumes no horizontal variability. The model is ideally suited to long-term investigations ranging from seasons to decades, and for coupling with biogeochemical models to explore the role that stratification and vertical mixing play on the dynamics of the lake ecosystem. GLM can be coupled with the AED2 for water quality simulations of lakes and reservoirs and integrated within a Markov Chain Monte Carlo (MCMC) algorithm. This allows for statistical analysis of model parameters to predict the dynamics of the lake system. A schematic of process considered in GLM is shown in Figure A.1.



Source: http://aed.see.uwa.edu.au/research/models/GLM/overview.html

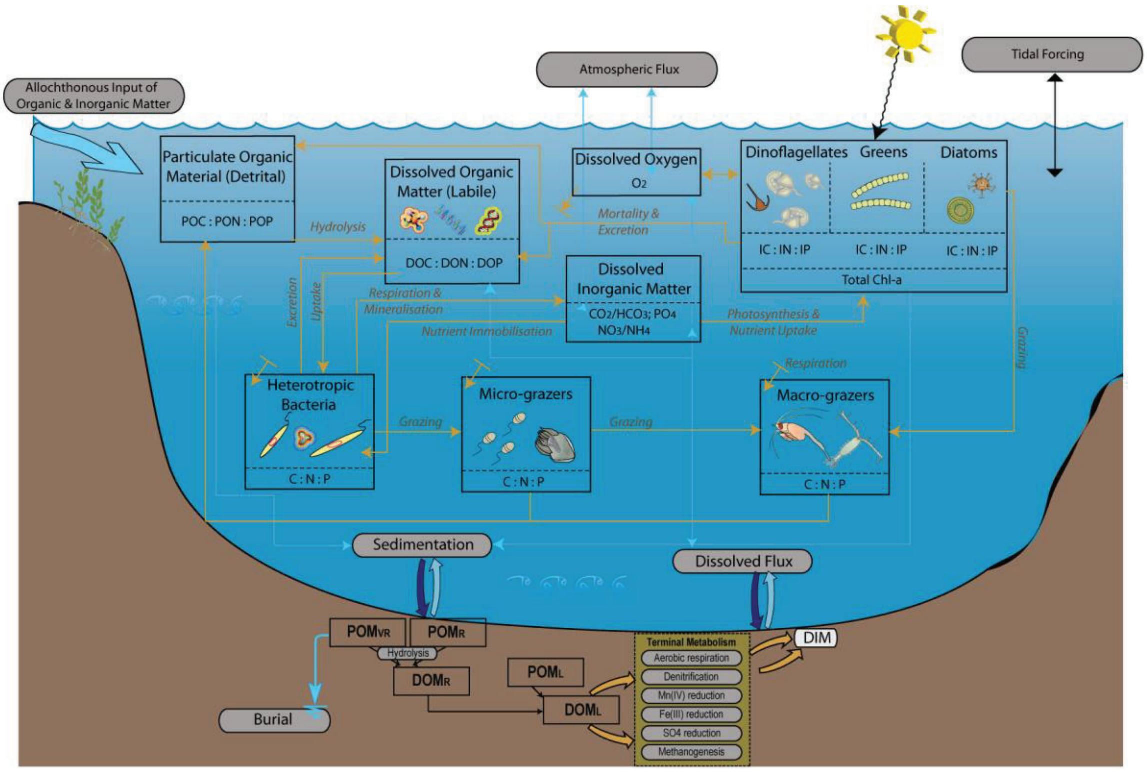
Figure A.1 Schematic of a GLM simulation with input information and key processes

Water quality model (AED2)

Water quality variables were resolved by the Aquatic EcoDynamics (AED2) model, developed by the AED group of the University of Western Australia. AED simulates relevant biogeochemical pathways to water quality, including nutrient and algal dynamics (Figure A.2).

In this study, the water quality model was used to resolve the nutrient cycling, uptake and growth of phytoplankton, as well as the notional uptake and excretion of bivalves. Advection and dispersion were supplied by the hydrodynamic model, GLM. The water quality model was also used to resolve the changes in dissolved oxygen in the water column. The specific suite of AED2 modules used in this study were:

- Dissolved oxygen;
- Nutrients (nitrogen, phosphorous and associated species);
- Organic matter (carbon, nitrogen and phosphorous, both particulate and dissolved);
- Marine diatoms;
- Bivalves as a sink and source of nutrients only (not their bio-energetics).



Source: http://aed.see.uwa.edu.au/research/models/AED/overview.html

Figure A.2 Aquatic ecosystem and nutrient processes simulated in AED2

PO Box 2305 Churchlands
WA 6018 Australia
Tel: +61 8 6163 4900
www.bmt.org