Stochastic effects of dredge plumes

Development and application of a risk-based approach to assess ecological effects of dredge plumes on sensitive receivers

E.J.C. Dupuits

- Original picture: http://www.sxc.hu/photo/1128319, modified to show seagrass and coral instead of spades and hearts.
- Original coral illustration: Joanna Woerner, Tracey Saxby, IAN Image Library
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Stochastic effects of dredge plumes – Development and application of a risk-based approach to assess ecological effects of dredge plumes on sensitive receivers

E.J.C. Dupuits

This thesis is submitted to Delft University of Technology in partial fulfillment of the requirements for the degree of Master of Science in Hydraulic Engineering

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Preface

This thesis is the result of my graduation project and concludes the Master of Science programme at the Faculty of Civil Engineering and Geosciences at Delft University of Technology. This study was initiated by Van Oord Dredging and Marine contractors bv. I would like to thank my graduation committee for their supervision, advice and support.

I would like to give out a special thanks to dr.ir. M. van Koningsveld, S.M. Doorn-Groen MBA, MSc, BSc, Dr. rer. nat. Elimar Precht and ir. Trine Cecilie Larsen for their valuable advices, comments and suggestions. Further I would like to thank colleagues from the environmental department of Van Oord for answering my questions and the nice working environment, and DHI Singapore for the nice working environment in Singapore. In particular, I thank my parents, family and friends for their interest and support.

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Summary

Dredge plumes and ecology have become increasingly intertwined in recent history; one of the more prominent reasons behind this is the influence of clouds of fine sediments released in the open water due to dredging ('dredge plumes') on for 'sensitive receivers' such as coral, seagrass and mangroves. An Environmental Risk Assessment (ERA) in combination with a cost-benefit analysis could be a valuable instrument for deciding on ecological impact. When dealing with ecology, it is important to realize that nature is not a harmonious, calm state: in fact, it could be considered to be at best only approximately in equilibrium, where small scale changes could lead to a completely new equilibrium. In the case of dredging operations and ecological effects, this is even more relevant because ecological effects occur constantly; not just due to dredging. An ERA consists of a description of the system and its components, hazard identification, effects assessment, exposure assessment, risk characterization and an evaluation which provides feedback for a possibly updated system description.

Globally following the steps in an ERA, exposures are looked at first. Relevant dredging exposures are suspended sediment concentrations and sedimentation. The source of this is the dredging equipment, which spills fine sediments in the open water. These sources lead to dynamic plume, which consists of interacting, complex three-dimensional fluid motions: sediment characteristics, bathymetry and dredging equipment type are just a few of the influencing factors. Because there motions are often still too complicated to model, the dynamic phase is captured in conservative estimates. These estimates serve as a source term for the passive plume phase. During the passive plume phase, advection and diffusion are dominant processes which are well understood processes. MIKE 21/3 or Delft3D, both hydrodynamic and transport models, can be used to schematize the passive plume.

Once a passive plume reaches a sensitive receiver, in this document assumed to be coral, seagrass or mangroves, effects can occur. It is important to realize that these effects do not have to be induced solely by dredging: other background effects can also provoke effects in sensitive receivers. Effects can be classified in discrete classes, ranging from growth to mortality. To which extent a receiver exhibits a certain effect depends on the receiver characteristics, the shape of a temporal variable exposure and the type of exposure. In an attempt to properly catch the effect of temporal variable exposures, several models are discussed with most prominent being the (originally toxicological) models of GUTS and ordered response model. Given their background these models appear to be limited to exposures which mimic a toxic exposure, although they are relatively easy to implement. At the other end of the spectrum are the energy budget models: models which try to describe important processes which together form an individual. While this is a more complete and holistic approach, there is concern regarding the uncertainty of the final answer given the large number of parameters (relative to the toxicological models) and the attributed uncertainties.

Combining these exposure and effect models in dredging practice and explicitly stating their (combined) uncertainties leads to a risk assessment for dredging practice. A part of this risk assessment was applied to field data regarding coral cover for a number of coral reefs, but the field data showed no strong correlation between exposure and changes in coral cover. Nonetheless, for a quantitative assessment of ecological risks, a probabilistic analysis is invaluable. This can lead to a proper valuation of the risk of effects, which in turn can be used in a cost-benefit analysis to properly weigh these risks if they are acceptable or not when compared to the benefits.

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Nomenclature

Roman Capital Symbols

$C_i(t)$	Internal concentration	$\left[M \cdot L^{-3}\right]$
$C_i^*(t)$	Scaled internal concentration	$\left[M \cdot L^{-3}\right]$
$C_e(t)$	External concentration	$\left[M \cdot L^{-3}\right]$
$D^*(t)$	Scaled damage level	$\left[M \cdot L^{-3}\right]$
$F_L(t)$	Arbitrary probability distribution function	$[-]$
P_f	Probability of failure for a system	$[-]$
P_i	Probability of failure for an element	$[-]$
$S_i(t)$	Individual survival distribution	$[-]$
$S_L(t)$	Arbitrary survival distribution	$[-]$
$V_{dredged}$	Volume dredged	$\left[m^3\right]$

Roman Symbols

$f_L(t)$	Arbitrary probability density function	$[-]$
$m_{fines;dry}$	Total mass of fines	$[kg]$
$frac_{fines}$	Fraction of fines	$[-]$
$f_z(t)$	Probability density function of a threshold for effects	$[-]$
k_a	Accumulation rate constant	$\left[T^{-1}\right]$
k_e	Elimination/dominant rate constant	$\left[T^{-1}\right]$
k_k	Killing rate constant	$\left[M^{-1} \cdot T^{-1}\right]$
k_r	Recovery rate constant	$\left[T^{-1}\right]$
$r(t)$	Hazard rate or failure rate	$\left[T^{-1}\right]$
$r_b(t)$	Background hazard rate	$\left[T^{-1}\right]$
$\rho_{dredged;dry}$	Dry mass of dredged material	$\left[kg/m^3\right]$

z	Threshold for effects	$[M^{-1}]$

Abbreviations

COMBO	Coral Mortality and Bleaching Output
CORSET	Coral Reef Scenario Evaluation Tool
CSD	Cutter Suction Dredger
DD	Degree Days
DEB	Dynamic Energy Budget
DHI	Dansk Hydraulisk Institut
DHM	Degree Heating Month
DHW	Degree Heating Week
DOM	Dissolved Organic Matter
ERA	Environmental Risk Assessment
GUTS	General Unified Threshold Model of Survival framework
HRF	Host Release Factor
IT	Individual Tolerance
LIT	Line Intercept Transect
PAR	Photosynthetically Active Radiation
PEC	Predicted Environmental Concentration
PNEC	Predicted No Effect Concentration
POM	Particulate Organic Matter
SD	Stochastic Death
SSD	Species Sensitivity Distribution
SST	Sea Surface Temperature
TASS	Turbidity ASsessment Software
TSHD	Trailing Suction Hopper Dredger
TSS	Total Suspended Solids
TKTD	Toxicokinetic-toxidynamic model

Chapter 1

Introduction

1.1 Ecology, risks and choices

Dredge plumes and ecology have become increasingly intertwined in recent history; one of the more prominent reasons behind this is the influence of clouds of fine sediments released in the open water due to dredging ('dredge plumes') on for example coral, seagrass and mangroves. As this is a field where many parameters come together, a risk assessment may be an attractive addition to the synthesis of ecology and dredge plumes.

1.1.1 Ecology

Ecology is according to German zoologist Ernst Haeckel who started using the term, the comprehensive science of the relationship of the organism to the environment (Haeckel, 1866). Contemporary books like Townsend et al. (2008) define ecology as the scientific study of the distribution and abundance of organisms and the interactions that determine distribution and abundance. Even though the word 'environment' is missing explicitly in the latter definition, it is actually still implicitly present in the term 'interactions'; Begon et al. (1990) defined the environment of an organism as all those factors and phenomena outside the organism that influence it, whether those factors be physical and chemical (abiotic) or other organism (biotic). The 'interactions' of Townsend et al. (2008) are the interactions with these factors; therefore the environment is still present (albeit implicitly) in his definition.

Becker (2011) noted in his master thesis, which shares the subject of ecological risk assessment for dredge plumes, the changing human perspective on the relation between (human) interventions in the environment, such as dredging, and the organisms that live in the environment. He described that in the past, results of interactions were seen as temporary; the course of history would mitigate and eventually dominate the interventions (Žižek, 2007). This should result in everything going on undisturbed, regardless of the actor's intention. Due to numerous advances in various fields, this could no longer be considered true. Interactions, or interfering activities, grew quickly in both size and number. Due to this development the result of interactions, be it beneficial or harmful, could no longer be considered temporary. Becker (2011) proceeded by observing that the awareness of lasting results of interactions, resulted in nature no longer being considered a historical constant. This knowledge has led to widespread discontent, and led to a movement set on opposing every change imposed by human activity. By opposing change due to human activity a balanced nature in a state of harmony and equilibrium should be achieved. This idea of nature contained and kept in equilibrium has largely been discredited by theories such as the Chaos theory (Kellert, 1994) and Catastrophe theory (Zeeman, 1976). These theories share the concept that systems are approximately in equilibrium. Small scale interventions result in fluctuations from some mean state, while sufficiently large interventions may lead to the establishment of a new approximate equilibrium. “As man is an integral part of the environment, neither man nor nature can be contained entirely. Pragmatic strategies should be undertaken in order to create sustainable development which is necessary to obtain an environment where man can safely survive” (Becker, 2011).

1.1.2 Motivation

But why should mankind bother with assessing (potentially devastating) environmental effects in the first place? Why should anyone care; what could be our *motivation*? The recent history of describing motivation in the field of motivational psychology is construed as reflecting two major metaphors: the person is a machine or the person is Godlike (Weiner, 1991). Both metaphors are containers for a number of theories which won’t be mentioned here; only the basic ideas behind these theories are treated.

Are persons mere systems or Godlike?

As mentioned in Weiner (1991), before Darwin presented his ideas a person was considered as a dualistic system; body and mind. The mind was exclusively attributed to humans, while subhumans (other living beings) only had a body. This difference was used to explain human behavior. Humans were created by God and in the image of God, having bestowed properties of God (by God). Therefore humans were considered Godlike. The body was seen as a source of desires that could be controlled by the mind. Subhumans on the other hand had no soul, and therefore no mind; they could not control their desires. Darwin proposed a mental evolution between humans and subhumans, leading to a view that we differ from subhumans but also share common traits. This meant the dualistic/monistic (with or without mind) separation between humans/animals could no longer be valid. Human behavior should, to some degree, be determined by reflexes and instincts. This leads to either subhumans and humans both being monistic (a being only has a body; a machine), or both being dualistic (a being has a mind and body) (Weiner, 1991). Although, for subhumans the mind must be equated with thinking and stripped of its association with the soul (Atkinson, 1964).

Making choices

These metaphors should not be taken literally; according to (Weiner, 1991) their value is more heuristic than analytical and more useful in the context of discovery than verification. Metaphors are not explanations. But, the body and its urges play a pivotal role in both the dualistic and monistic system. If the theory of Darwin is used and accepted, humans evolved from subhumans; which means we share some of the fundamental features of all known life: the need for reproduction and self-preservation. Either the body, or the combination body and mind has to respond to the callings of these fundamental features; decisions are then made based on options presented. Naturally there are more higher order features influencing decision making (think of phenomena such as duty, or avoiding pain), but those are left unexplored here. Intricacies of the nature of making decisions are depicted wonderfully in the fourth verse of the Aeneid by Virgil (Greenough, 1900), and its (loose) English translation by Virgil and Dryden (1997). Aeneas, the Trojan hero of the story, fled the burning city of Troy with his people and is in search of new land to start a new enclave. He was ordered to do so by the gods, but during his journey he came across Dido, the queen of Carthage. They fell in love, but eventually Aeneas decided to leave and continue his journey. The following describes what goes on in the body (and mind?) of Aeneas just after he told Dido of his intentions to continue his journey:

Talibus orabat, talisque miserrima fletus
fertque refertque soror: sed nullis ille movetur
fletibus, aut voces ullas tractabilis audit;
fata obstant, placidasque viri deus obstruit auris.
Ac, velut annoso validam cum robore quercum
Alpini Boreae nunc hinc nunc flatibus illinc
eruere inter se certant; it stridor, et altae
consternunt terram concusso stipite frondes;
ipsa haeret scopulis, et, quantum vertice ad auras
aetherias, tantum radice in Tartara tendit:
haud secus adsiduis hinc atque hinc vocibus heros
tunditur, et magno persentit pectore curas;
mens immota manet; lacrimae volvuntur inanes.

As, when the winds their airy quarrel try,
Justling from ev'ry quarter of the sky,
This way and that the mountain oak they bend,
His boughs they shatter, and his branches rend;
With leaves and falling mast they spread the ground;
The hollow valleys echo to the sound:
Unmov'd, the royal plant their fury mocks,
Or, shaken, clings more closely to the rocks;
Far as he shoots his tow'ring head on high,
So deep in earth his fix'd foundations lie.
No less a storm the Trojan hero bears;
Thick messages and loud complaints he hears,
And bandied words, still beating on his ears.
Sighs, groans, and tears proclaim his inward pains;
But the firm purpose of his heart remains.

The argument whether this is a depiction of a dualistic or monistic system is not of interest here (either could be argued to be true); neither is it in finding his drive(s) which lead him to this decision. The importance of choice and its grounds is shown here. Aeneas basically had two choices; one option would be to start a family with Dido and stay in Carthage, or he could continue his journey. His first option would have a predictable future, the second could have had wildly different outcomes. However, Aeneas was assured by the gods that he eventually was bound for greatness by founding the city of Rome. If the gods, which in this story are seen as the authoritative figures on everything fate related, did not bother with telling him of the outcome of his future journey it could be argued that the uncertainty related to the outcome of his future journey would have been substantially larger; thus his choice *might* have been rather different.

Weighing options

Caring for our environment could be explained as being motivated to make certain decisions. As the well-being of our environment influences our capabilities of self-preservation and reproduction (positive or negative), we are motivated to make a decision if the consequences of an interaction with our environment can be shown. But, as shown in the story of Aeneas, making that choice is mostly based on having relevant information describing possible outcomes of the options presented. If there aren't any gods available disclosing this kind of information, we should still have information that describes outcomes and the surrounding uncertainty.

1.1.3 Risk assessment

Risk assessment provides such a method in which possible outcomes are described, even if significant factors are uncertain. “The particular value of risk assessment is that it demonstrates how prior knowledge revisited within the context of an accepted framework provides a clear exposition of technical assumptions, their application within an analytical approach, and an explanation of the attendant uncertainties” (PIANC, 2006).

PIANC (2006) defined risk assessment as “a formalized basis for the objective evaluation of risk that clearly presents and considers assumptions and uncertainties. Risk assessment commonly involves multidisciplinary efforts among physical scientists, toxicologists, biologists, mathematical modelers, and information specialists.” A specific version of risk assessment is the Environmental Risk Assessment (ERA). According to PIANC (2006) this is the process of quantitatively evaluating the effect of an exposure (e.g. a chemical or physical condition) upon the health of individual humans or the environmental well being of an individual, population or community of animals and plants. The former is called human health risk assessment, the latter ecological risk assessment. Risk is quantified as risk exposure or the potential for some other effect as a comparison of a predicted exposure level to a scientifically established benchmark level. “Risk assessment is the only approach currently available for quantifying risks that has broad acceptance in the scientific and regulatory communities. It is not perfect, but it is the most logical, technically sound approach to objective evaluation of potential environmental impacts presently available” (Bray, 2008).

ERA is a great aid for decision making. However, decision makers use ERA mostly to decide if a risk is unacceptable or acceptable (Bray, 2008). While this correctly implies that some adverse effects might be deemed acceptable, it does carry a negative attitude. Possible beneficial effects due to proposed interventions should not be forgotten! Unfortunately, in practice emphasis is too often placed on detrimental effects. In order to make a good cost-benefit analysis, all effects should be included; both adverse and beneficial effects. The value of either beneficial or adverse ecological effects however is problematic and controversial at best, which leads to problems when trying to compare the value of a project with the costs or benefits of its effects. However, lacking the ability to quantify an effect properly should not be a reason to omit the effect entirely; a qualitative description should still be present.

1.1.4 Making the right choice

Subsection 1.1.2 described why we should care for our environment, while Subsection 1.1.3 gave a formalized tool for weighing options. It is not too far fetched to conclude from these subsections that, given a crystal clear risk assessment, making the right choice for the environment and ourselves should be obvious. Reality is, unfortunately as always, much more complicated than that. Even if the notions of a ‘right’ choice and a perfect risk assessment are omitted from discussion, an individual could still reach a different conclusion what would be the better choice than suggested by a risk assessment. An example of such a deviation can be found in “Tragedy of the Commons” (Hardin, 1968).

In “Tragedy of the Commons” the argument was made that “The population problem has no technical solution; it requires a fundamental extension in morality” (Hardin, 1968). (The population problem is the growth of the global population in light of limited resources and the manner we use these resources.) In this article, the problem presented is the notion of commons (several years later rectified by the author to *unmanaged* commons) and the use of it by individuals: if an individual increases his use of a common, only he gets the gain of the increased usage, while the burden on the common due to the increased usage is shared by all users of the common. Even if an individual is fully aware that his actions could eventually lead to the decline or even destruction of the common, this individual will continue to maximize his gains because the gains continue to outweigh the (shared) burden. "Each individual is locked into a system that compels him to increase his gain without limit - in a world that is limited (Hardin, 1968)."

If only a few individuals use a relatively large common, no lasting effect is expected and the common can be approximately be seen as infinite. This is supported by, for example, in the 1609 work of *Mare Liberum*. In those days, the oceans were considered to be of such scale that it suffices for any possible use on the part of all peoples for drawing water, for fishing, and for sailing (Grotius et al., 2004). Presently, that notion cannot be seen as true anymore; as was mentioned in Subsection 1.1.1, recent history saw effect by mankind greatly increase in both size and number which makes the environment, and the commons it harbors, finite and limited. Considering the oceans a-priori as sufficient for all is therefore a mistake. One popular argument is the fact that technology can also lessen the burden we place on commons. To a certain extent, technological improvements can do this - but they are bandages for the described morality problem, as they only delay the destruction of a common, and do not offer an integral long term solution.

The individual gets to maximize his/her gain, while society - of which he/she is part of - suffers as a whole; the problem is the freedom of individuals in unmanaged commons. The solution as proposed in "Tragedy of the Commons" is mutual agreed upon coercion (Hardin, 1968). Mutual agreed coercion is hardly something new, there are numerous examples in past and present of society imposing limits on the freedom of individuals: For example think of laws prohibiting us from robbing banks, or having to pay taxes. Taxing the usage of commons is already used in practice in for example CO_2 certificates, which give the right to emit a certain amount of CO_2 into the air (which is an unmanaged common) - instead of just emitting CO_2 to the liking of the individual emitter.

The adjective 'unmanaged' is quite important for the "Tragedy of the Commons": without it, 'commons' could include managed commons, or previously managed commons. Even though the latter indicates a common is presently unmanaged, the management it had in the past makes it a more complicated, and possibly unfit, case for the "Tragedy of the Commons". For the synthesis of risk assessment and dredge plumes, the world's oceans can be considered as an unmanaged common. The assumption that the oceans are indeed finite seems a plausible starting point, considering current problems in and around the world's oceans, and limiting freedom in the usage of unmanaged commons is (thus) a necessity (Hardin, 1968). Education can play a big part in elucidating the reasons behind the necessity for society to limit the freedom of individual usage of commons, and a good candidate for a tool in explaining this is the aforementioned risk assessment in Subsection 1.1.3. However, the relation between education and risk assessment could be better described as symbiotic. Education can help gain more insight in processes risk assessment describes, while the descriptions in a risk assessment could also lead to better understanding of processes, which could advances understanding and education. Any existing coercion can help to quantify effects in risk assessments (emitting amount x of CO_2 costs y dollars), although the coercion itself could (or should?) be assessed too: “Is the coercion indeed commonly agreed upon?”, “can it be shown to be an effective measure?”, and “is the measure actually required?” are examples of questions that should be answered.

1.2 Thesis

Ecology and dredge plumes are intertwined because dredge plumes are assumed to have an effect on parts of ecological structures. Formally put: exposure stemming from dredge plumes affect receivers. The group of receivers commonly assumed to be sensitive for these exposures consists of coral, seagrass and mangroves; these receivers are also called ‘sensitive receivers’.

1.2.1 Objective

Current dredging practice still approaches ecological aspects from a precautionary principle: even if cause and effect are not fully established, precautionary measures are required when ecological impact is suspected. Being overcautious is better than being reckless when risks are not known; but predicting the risks and explicitly stating their uncertainty/variability (by using a risk assessment) leads to a more complete overview. This can still lead to a conservative approach if required. The previous sections conveyed that a probabilistic risk assessment is a more insightful and thus preferred method as opposed to the precautionary principle. Proposed here is to establish cause and effect, express uncertainties explicitly and interpret the resulting answer. In the master thesis of Becker (2011), a framework was given for practical applications of ERA in current dredging practice. The intent of this master thesis is to delve deeper into the proposed layout of this framework, and various parts of this framework in order to bring an actual practical application closer to reality. This intent leads to a similar objective as in the master thesis of Becker (2011):

> **Development and application of a risk-based approach to assess ecological effects of dredge plumes acting on sensitive receivers**

Research questions

Given the existing master thesis of Becker (2011) on this subject and its recommendations, and the research objective in this thesis, the following questions are posed using Figure 1.1 as a guideline; the events in this figure will be further explained in Chapter 2.

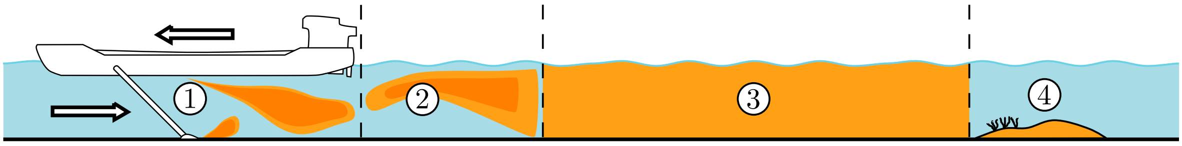

Figure 1.1: Overview of a dredging operation, encompassing four key events from dredger to sensitive receiver.

I *Event 1 and 2:* How do dredging related activities (sources) contribute to exposure signals, and how can these be quantified?

II *Event 3:* How do exposure signals stemming from dredging activities develop, and how can these be quantified?

III *Event 4:* Which aspects of exposure signals are relevant for quantifying exposures for sensitive receivers?

IV *Event 4:* How can effects, concerning sensitive receivers, be quantified?

V *Overall:* How can data concerning exposure and effect be gathered and applied (e.g. in models) for sensitive receivers?

VI *Overall:* How can a risk-based approach be applied to current dredging practice?

As the the research questions suggest, much of the focus in this thesis goes to event 4, and to a lesser extent event 3, of Figure 1.1 which will result in more extensive treatment of these subjects.

1.2.2 Approach

Following the steps in Figure 1.2, a general overview of dredging practice and a probabilistic approach is given in Chapter 2. Connected to a probabilistic approach in dredging practice, the terms uncertainty, source, exposure, effect and sensitive receivers are considered. In Chapter 3, a description is given of sources and the various stages eventually leading to an exposure which can be used for an exposure-effect relation. Uncertainties regarding the transformation from source to exposure are also briefly mentioned.

In Chapter 4, an assessment is made of effects: what is the influence of a sensitive receiver on the effect, how can effects be measured and quantified, and how can exposure-effect relations be characterized. A number of schematizations describing exposure-effect relations are treated.

Once exposure and effect have been assessed, the findings are connected to parts of a probabilistic method for dredging practice in Chapter 5. An attempt to actually use the described exposure-effect models can be found in Chapter 6; findings of this and the previous can be found in the conclusions and recommendations in Chapter 7.

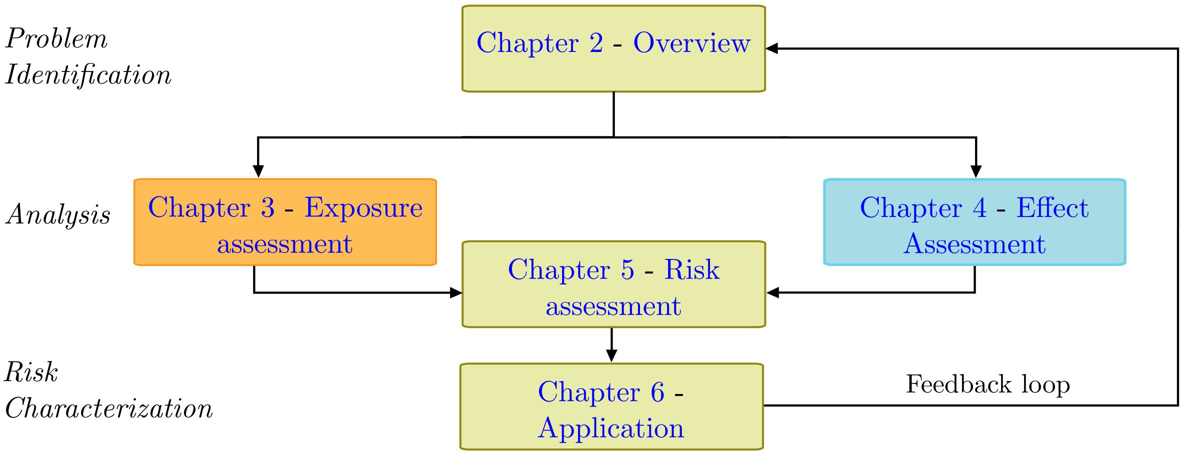

Figure 1.2: Global layout of this thesis, which was inspired by the steps proposed in an ERA.

Chapter 2

Overview

2.1 Dredging practice

The illustration from Figure 1.1 is shown again in Figure 2.1 which serves as an overview of important events in a dredging operation. The following four key events are shown:

1. *Around the dredger:* Soil is dredged, which results in fines being released in the open water. Part of this is due to the draghead of the TSHD (Trailing Suction Hopper Dredger), but most of the fines are generated by overflowing water coming out of the TSHD.
2. *Relatively close to the dredger:* In this part of the sequence, a lot of complex processes take place when the relatively dense overflow water moves to the bed as a density current (the dynamic plume). Fines that do not go into the plume settle to the bed.
3. *Relatively far away from the dredger:* At the start of the so-called far field, the sediment plume is no longer affected by the ship. This is also called the passive phase: only convection, diffusion and precipitation affect the concentration of the sediment plume.
4. *Arrival of sediment plume at sensitive receiver:* If a sediment plume reaches a sensitive receiver, depending on the properties of the sediment plume (concentration levels, exposure duration, sediment type etc) and the type of sensitive receiver and its properties (state of health, resilience to reduced light condition and sedimentation effects, etc) a certain environmental effect may occur.

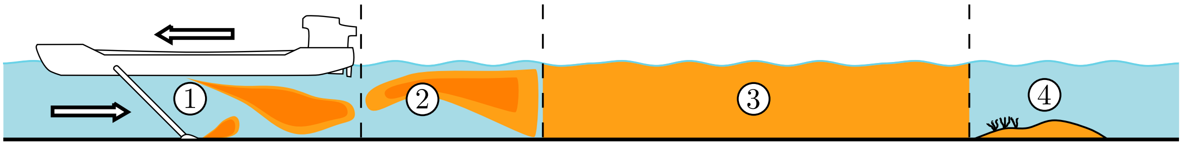

Figure 2.1: Overview of a dredging operation, encompassing four key events from dredger to sensitive receiver.

2.2 Probabilistic approach

Dredging is considered to be the cause of a load (the dredge plume) on a sensitive receiver. The effect on the sensitive receiver depends on the magnitude of the load, and the resilience (strength) of the sensitive receiver. This can be captured in a probabilistic assessment. In its most general and simple form, a probabilistic calculation is a simple comparison between a 'Load' variable (S) and a 'Strength' (R) variable. These variables are commonly expressed as stochastic variables, also known as random variables: the name implies that these variables have a value with a certain variability. The reliability function Z in equation [2.1] indicates failure or 'unreliability' when the load exceeds the strength; this occurs when Z becomes smaller than zero ($Z < 0$). The associated variability and the influence on the reliability function Z is visually depicted in Figure 2.2. The overlap between load and strength indicates the probability of failure.

$$\begin{gathered} Reliability = Strength - Load \\ Z = R - S \end{gathered} \tag{2.1}$$

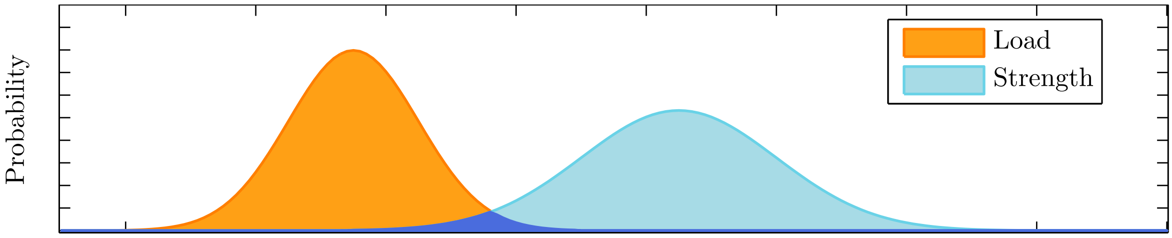

Figure 2.2: Fictitious Load (S) and Strength (R) stochastic variables. The overlap between S and R indicates the probability of failure of (for example) a sensitive receiver.

Even though 'Load' and 'Strength' are often used terms in probabilistic analysis, in the current context of dredge plumes interacting with sensitive receivers the words 'Exposure' and 'Effect' respectively would be more appropriate. These substitutions emphasize the more general nature of sensitive receiver effects: an effect resulting from an exposure does not always have to be of a negative nature, a positive effect is possible as well. After all, in essence the function in [2.1] shows that if the resistance or threshold to a certain effect is exceeded by an exposure, the effect has a chance of occurring: if one feeds an organism sufficiently, it will probably grow. If an organism is cut often and deep, it will probably die. Growth or mortality are just two of the many forms 'Failure' in equation [2.1] can take.

2.3 Uncertainty

The variability in stochastic variables stems from uncertainty. According to Van Gelder (2000), there are two categories of uncertainties: inherent uncertainty stemming from randomness or the variations in nature, and epistemic uncertainty caused by lack of knowledge of all the causes and effects in physical systems or lack of sufficient data. Epistemic uncertainty includes both model limitations and statistical uncertainty (parameter and distribution uncertainty). Two specific variants of epistemic uncertainty warrant some extra attention: the influence of human actions and modeling uncertainty.

2.3.1 Human actions

Separation of human and modeling uncertainty has been suggested (among others) by De Corn and Inkabi (2010). Especially for processes significantly influenced by human actions, this separation is useful. This is relevant for this thesis because the strength of sensitive receivers is commonly quantified by manual inspection of for example photographs of coral cover. Human actions, even though they have partial inherent uncertainty, are epistemic.

Human reliability is influenced by a large number of factors including, but not limited to, the complexity of the task, previous training and time available to perform the task. Various studies have been performed in the past, for example in Bea (1997) or Gertman and Blackman (1993), looking to quantify various general tasks executed by humans. A result from such studies can be seen in Figure 2.3. Ideally, a complete study should be performed to asses the likelihood of failing at tasks specific to the problem at hand, but for a first estimate the data in Figure 2.3 can be useful. The data in this figure indicates that there is a large spread related to the likelihood of failing at a certain task. This also means there is room for optimization if a task has a relative high likelihood of failing.

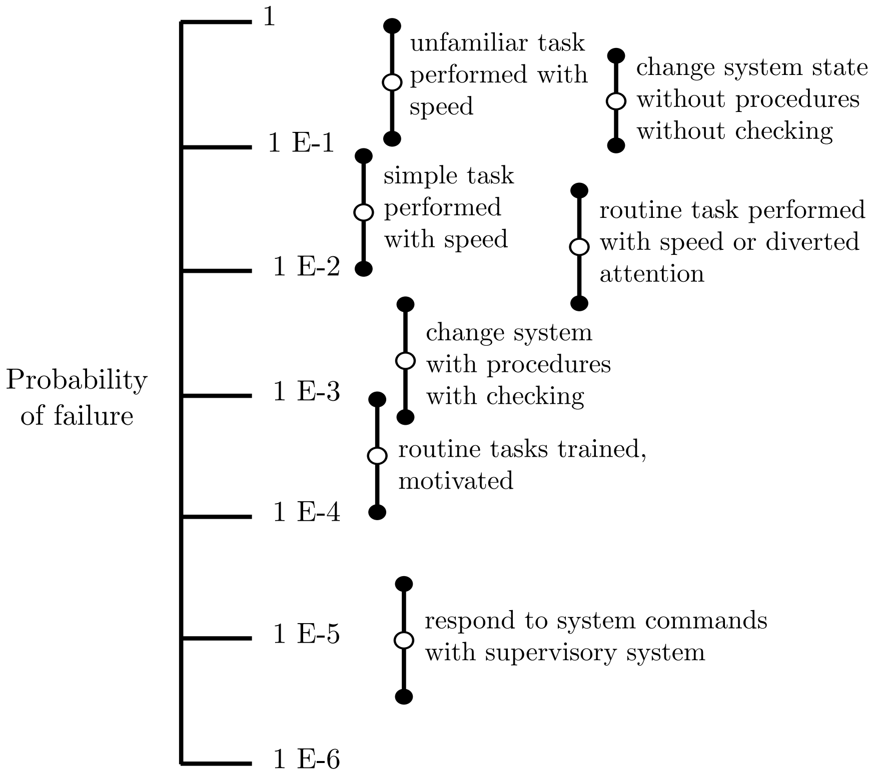

Figure 2.3: Nominal human performance task reliability. Adapted from De Corn and Inkabi (2010); Williams (1988).

2.3.2 Modeling uncertainty

Modeling uncertainty commonly brings forth thoughts of limitations, simplifications and assumptions in a model. These uncertainties are often well documented for each model on its own and the answer is usually given with certain confidence: value X is not exceeded with Y% confidence (95% is a commonly used confidence number). Confidence intervals are a common way to indicate the variance of a stochastic variable when comparing to e.g. the mean value. If a stochastic variable has a certain distribution, the 95% confidence interval covers 95% of the distribution. This cover can represent the distribution excluding 2.5% on the left and right tail (resulting in two representative numbers), or the distribution excluding 5% on either the left or right tail (resulting in one representative number). The latter is shown in Figure 2.4. The choice between representations depends on what the value represents.

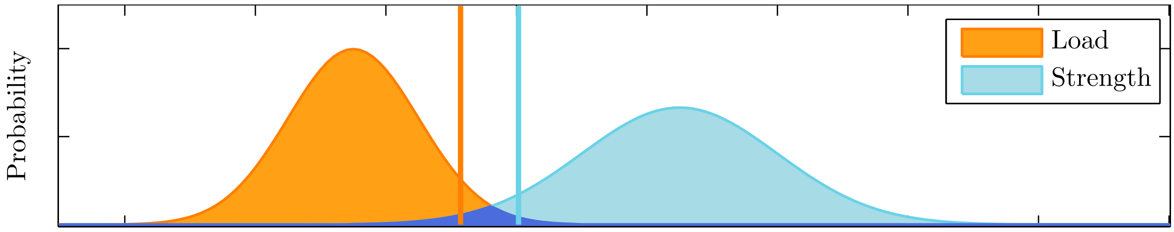

Figure 2.4: 95% confidence intervals for Figure 2.2. For the 'Load' stochast this equals 95% non-exceedance interval, for the 'Strength' stochast it equals the 95% exceedance interval.

The four steps shown in Figure 2.1 could also be seen as four critical steps in the calculation of an effect due to dredging, which are usually calculated separately. However, each of these four steps is used successively in reaching a correct effect assessment. This is shown in a calculation event tree in Figure 2.5. This means that multiple successive steps, each giving an answer using 95% confidence, result in an ever smaller final confidence percentage. In order to reach a final global desired confidence, higher local confidence levels should be attained in the sub steps. It is stressed that the number of steps are just another factor in introducing uncertainty but certainly not the only factor.

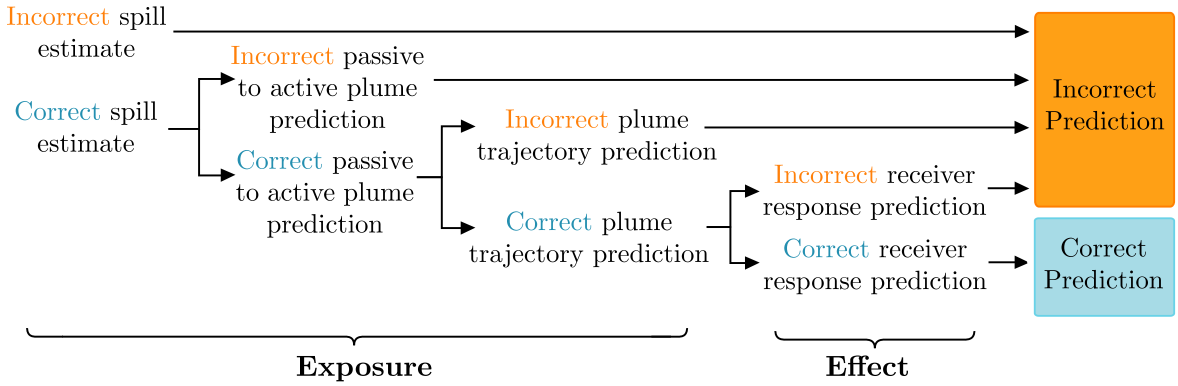

Figure 2.5: Flow scheme representing a global calculation scheme, based on Figure 2.1. Because each successive step uses data from a previous calculation round, uncertainty propagates through the scheme and increases overall uncertainty.

If the event tree in Figure 2.5 is seen as a model of effects on sensitive receivers due to a dredge plume, a number of uncertainties can be distinguished. Models used in each step usually contain simplifications, which introduces model uncertainty. Natural variations in for example wind speed or currents leads to the introduction of inherent uncertainty. If a model is used for determining the effect on sensitive receivers, there is parameter and distribution uncertainty due to parameter fitting on limited number of data.

The event tree in Figure 2.5 indicates that, in order to reach a correct value for the load, three load calculations have to be made. If in each of these three steps a small error is made, these errors will propagate throughout the calculation set. To get an idea of the influence of errors in each step, the three steps can be seen as series system; each step needs to be calculated correctly in order to reach a correct value for the load. Simple calculation rules for series and parallel systems are given in Table 2.1.

	Components		
System	**Mutually exclusive**	**Independent**	**Dependent**
Series	$\sum_{i=1}^{n} P_i$ (Upper bound)	$1 - \prod_{i=1}^{n} (1 - P_i)$	$\max\{P_i\}$ (Lower bound)
Parallel	0 (Lower bound)	$\prod_{i=1}^{n} P_i$	$\min\{P_i\}$ (Upper bound)

Table 2.1: Theoretical failure probability upper and lower bounds for series and parallel systems (Vrijling et al., 1997).

Because events are not mutually exclusive (the occurrence of event 1 does not exclude the occurrence of event 2 or 3), the upper bound for the failure probability is the assumption of independence of components. If each step has an assumed 95% chance of success (and therefore a 5% chance of failure due to uncertainties or errors), the chance of failure increases to approximately 15% for the combined result of all three steps:

$$P_f = 1 - \prod_{i=1}^{n} (1 - P_i) = 1 - \prod_{i=1}^{3} (1 - 0.05) \approx 0.143$$

Which also implies that the chance of success is about 85%. Again, these confidence intervals should not be confused with the accuracy of the answer: confidence intervals only indicate how much confidence is placed in the given answer, not how good the answer is. In fact if the model calculation is rather crude, the variance of the answer is often large. This means the $Y\%$ confidence interval is also large, which results in a conservative answer. The confidence placed in this answer might be high, however the answer itself might not be accurate.

2.4 Source, exposure and effect

In this thesis the direct removal of sensitive receivers by means of executing a dredging operation at the location of sensitive receivers is not discussed. Focus is placed on dredge plumes; though it is worth mentioning that exposing sensitive receivers to a dredging operation directly would have very serious consequences, possibly magnitudes larger than consequences resulting from dredge plumes.

Dredge plumes are in fact sediment plumes originating from dredging activities. This implies that seeing a *sediment* plume does not always mean it is in fact a *dredge* plume. Sediment plumes can originate from propeller wash of passing ships, fishing activities, river run-off or re-suspension during storms. Figure 2.6 further clarifies that multiple sources can lead to sediment plumes: Multiple sources can result in multiple, different exposures, which in their turn can lead to multiple effects; though multiple sources sources could also lead to very similar exposures. In the case of a dredge plume the source is dredging equipment, exposures could be reduced light and sedimentation, while the effect could be mortality of coral. By now terms such as exposure and effect have been frequently used, and are described in Table 2.2.

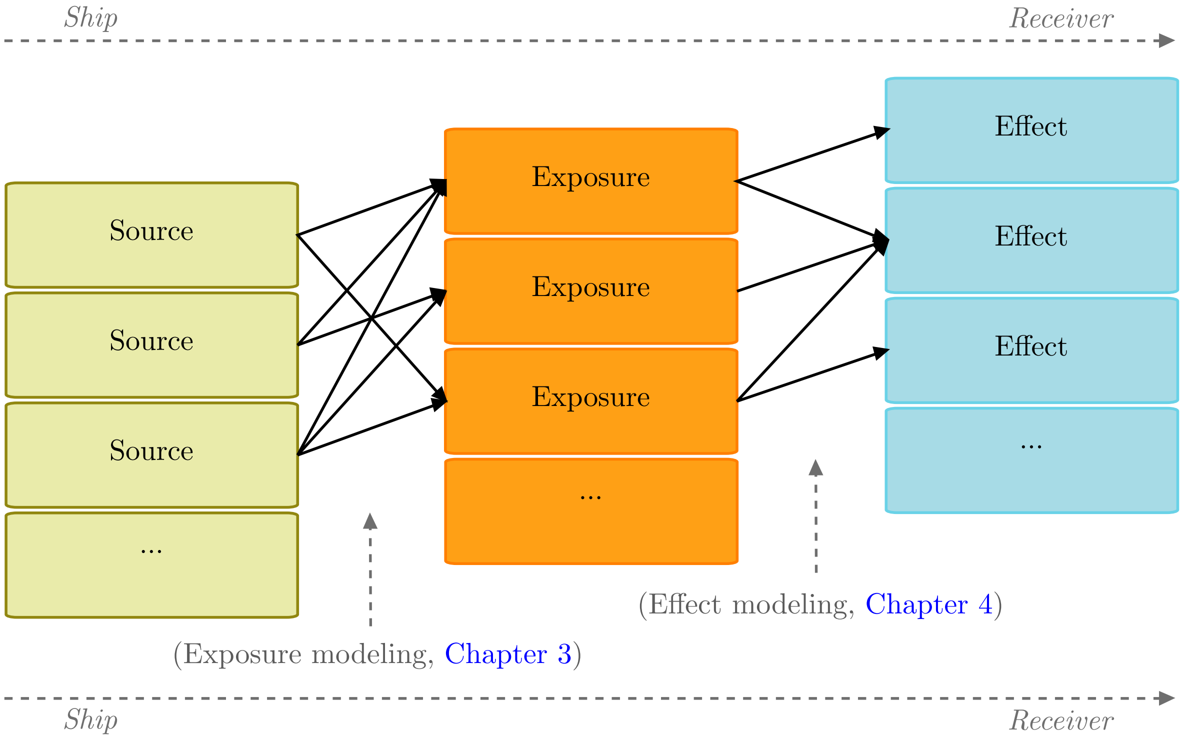

Figure 2.6: Graphical representation noting that multiple sources can result in multiple (different) exposures, which on their own could lead to multiple effects. In the case of a dredge plume, the source would be dredging equipment.

2.4.1 Factors influencing exposure

Besides other sources or even background (floating) sediment having the possibility of being more significant than dredge plumes on their own, the presence of dredge plumes can (to a certain extent) be controlled by taking into account matters like currents, tides, wind speed and direction. The implementation of silt screens could, potentially, also significantly reduce

Term	Description
Dose	Total amount of an agent administered to, taken up by, or absorbed by an organism, system, or (sub)population.
Dose-response relationship	Relationship between the amount of an agent administered to, taken up by, or absorbed by an organism, system, or (sub)population and the change developed in that organism, system, or (sub)population in reaction to the agent.
Source	Origin held accountable for one or multiple exposures.
Effect	Change in the state or dynamics of an organism, system, or (sub)population caused by the exposure to an agent.
Exposure	Concentration or amount of a particular agent that reaches a target organism, system, or (sub)population in a specific frequency for a defined duration.
Impact	An adverse effect.
Hazard	Inherent property of an agent or situation having the potential to cause adverse effects when an organism, system, or (sub)population is exposed to that agent.

Table 2.2: Key generic terms in an Ecological Risk Assessment (ERA) and their definitions (WHO, 2004), as found in Becker (2011) with the additional entries for impact and source.

dredge plumes. The last option is not extensively discussed in this thesis, because the goal of this thesis is to assess effects of a dredging operation on sensitive receivers. Alternatives like silt screens can only be thoroughly reviewed once a baseline, i.e. effect of dredge plumes without silt screens, is established.

Other sources for increased sediment concentrations are the aforementioned propeller wash of passing ships, fishing activities, river run-off or re-suspension during storms: a number of these sources are shown in Figure 2.7. In Lake Michigan and the Mississippi River, after storm events, suspended sediment concentrations in the range of 15-30 mg/L were measured: these can be considered to be roughly in the same order as some dredging-induced concentrations (Aarninkhof, 2008); even though dredging induced suspended sediment concentrations can be significantly lower or higher than this range.

Seasonal variability in turbidity has been shown by e.g. Shi and Wang (2010). This means turbidity changes in space and time. A rough division of turbidity was proposed by Shi and Wang (2010): clear water with $0 < K_d(490) \leq 0.1m^{-1}$, modestly turbid water with $0.1 < K_d(490) \leq 0.3m^{-1}$ and turbid water with $K_d(490) > 0.3m^{-1}$; where $K_d(490)$ is a diffuse attenuation coefficient at wavelength 490nm. This can be used to interpret Figure 2.8. This leads to a more qualitative, but still useful insight in global turbidity variations. Significant seasonal variability was shown, especially for modestly turbid waters. Variability in the open ocean was credited to dominating seasonal phytoplankton bloom (clear water and modestly turbid water), while on the other hand for the coastal region turbid waters were attributed mostly to sediment concentrations (Shi and Wang, 2010).

(a) Sediment plume originating from river run-off. Taken from http://earthobservatory.nasa.gov/IOTD/view.php?id=1257; original source Liam Gumley, Space Science and Engineering Center, University of Wisconsin-Madison and the MODIS science team.

(b) Sediment can also be re-suspended due to ship traffic. Image copyrighted by Kevin Harber, found at http://www.flickr.com/photos/kevharb/3608833836/.

(c) TSHD dredge plume. Image found on http://publicwiki.deltares.nl/display/BWN/Building+Block+-+Assessment+of+dredging-induced+turbidity, no original source could be found.

Figure 2.7: Examples of sources which could contribute to increased suspended sediment concentrations. The scale of each picture varies significantly.

2.4.2 Factors influencing effect

As seen in Figure 2.6, exposure leads to effects: in a more formal and rather abstract way it could be said that if exposure x exceeds the corresponding exposure threshold for effect y, effect y will occur. But in practice, the severity, duration and even threshold of effect y could be different for individuals, communities, races or species. So if the exposure would be exactly the same for each, what determines the type and severity of the effect?

Type and severity of an effect

On an abstract level, effects are descriptions of parts of the development and/or behavior of an individual. Ideally a model describing the the whole development of an arbitrary individual would exist. Unfortunately describing effects accurately in practice seems to be a problem already, with only a small subset having extensive and quantitative literature cover. However, given a relevant set of effect categories one could achieve a satisfactory approximation of the whole development/behavior of an individual. In order to achieve this, at least one positive and one negative effect need to be described. The actual categorization of effects depends on practicality (i.e. available data) and the specimens in question.

Effects are typically ordered: the set of conditions applicable for a certain effect will get smaller for an ever more increasing positive effect. A similar thing can be said about negative effects: the set of conditions will get smaller for an ever more increasing negative effect. The most positive effect generally accepted for a given species is when it ‘performs best’, that

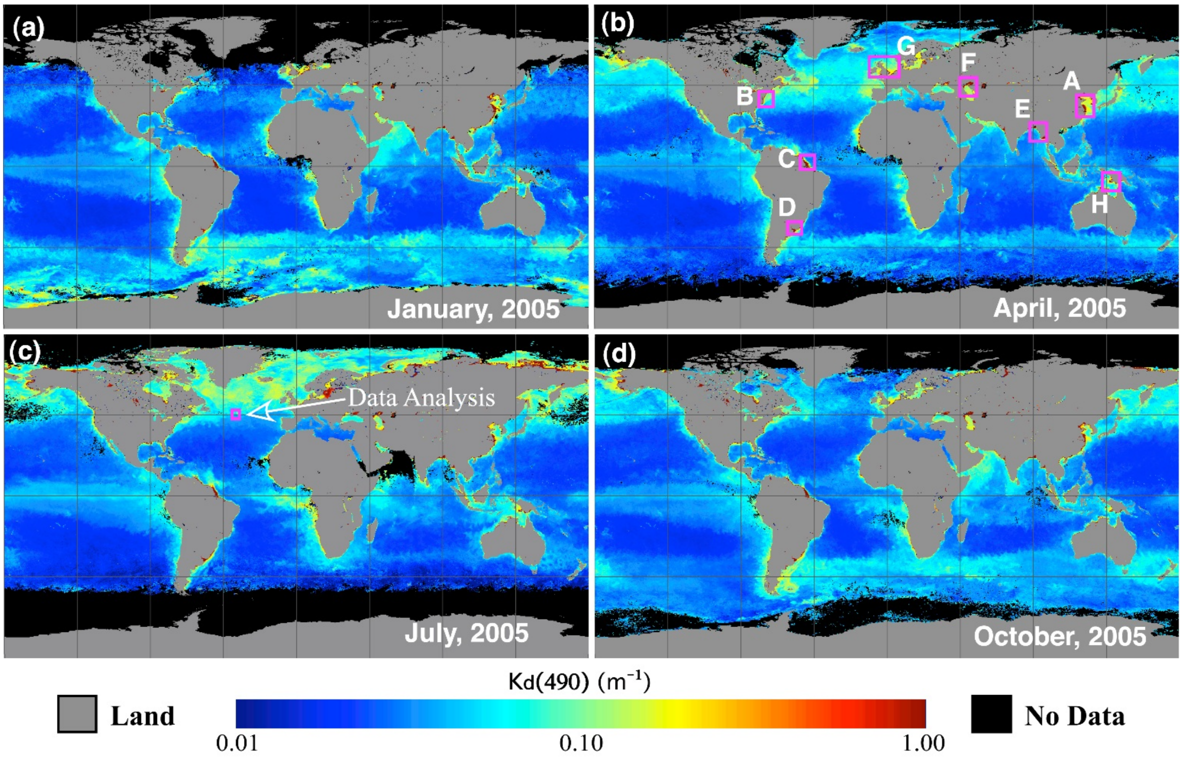

Figure 2.8: Seasonal distribution of global water turbidity as represented with $K_d(490)$ in 2005 during the periods (a) January (winter), (b) April (spring), (c) July (summer), and (d) October (autumn). Eight major turbid regions are marked in (b). Description and image taken from Shi and Wang (2010).

is when individuals of a species leave the most descendants. The commonly accepted most negative effect is mortality.

Individuals are not the only point of interest. Individuals of the same species form populations, and a number of populations form a community. While the most positive effect generally accepted for an individual is optimal reproduction, additional effects might be considered optimal (or most positive) for populations or communities. For example, for a community this might be diversity or preventing erosion: the interests of the environment start playing a larger role - of which we humans are very much part of. Populations and communities also add complexity by means of respectively intra- and interspecies behavior: this could be for example complementary, substitutable, inhibiting or antagonistic relationships (the first two having a positive influence, the latter two a negative influence) (Begon et al., 1990). If one or more populations change in size in a community while other populations remain relatively unaffected, the whole community could change dramatically with respect to, for example, composition or size. A more extreme case of this is he entire collapse of a community, because key accommodating populations were lost. Newell et al. (1998) described such an event for certain communities.

Conditions are defined as abiotic environmental factors which vary in time and space, and to which organisms are differentially responsive, i.e. salinity, flow velocity and concentration of toxicants (Begon et al., 1990). Another important factor defining effects are *resources*:

all things consumed by an organism; or quantities that can be reduced by the activity of the organism. Resources have a broad definition, and also includes light (strictly speaking: solar radiation) as a potential resource. Radiant energy is converted during photosynthesis in energy-rich chemical compounds of carbon. These compounds are subsequently broken down in respiration either by the plan itself or by those that eat to decompose it (Begon et al., 1990). Even though solar radiation in general can be seen as an infinite resource on the timescale of the average lifespan of an organism, the amount of solar radiation at a specific location is finite because an organism needs a surface to capture this radiation, preventing other organisms to capture the radiation at the very same surface. This is an example of unsystematic variation of radiation; systematic variations are the diurnal and annual rhythms of solar radiation. Seasonal variations in radiation are much more pronounced at higher latitudes than around the equator (Begon et al., 1990).

Temperature is an example of an important condition. Variations in temperature can also be divided in unsystematic and systematic variations. In general, plants have a higher metabolism rate at higher temperatures; this relation holds until a certain optimum is reached. Above this optimum respiration is faster than photosynthesis can keep up with, effectively starving the plant; though in practice the indirect effect of dehydration is a main cause of mortality. On the other hand, low temperatures can lead to slow or stopping metabolic reactions, or even freezing. However, if there is systemic seasonal variation regarding temperature, a plant could adjust (or 'harden') to survive at higher or lower temperatures then it would be able to withstand in a different season (Begon et al., 1990).

2.5 Sensitive receivers

The group of receivers commonly assumed to be sensitive for dredging exposures, and which is treated in this thesis, consists of coral, seagrass and mangroves; these receivers are also called 'sensitive receivers'.

2.5.1 Mangroves

Mangroves thrive in very dynamic situations and can be found primarily in the upper tidal zone. Due to anaerobic conditions in the muddy soil, exchange with the atmosphere is improved by means of aerial roots (Schiereck, 2000). The two principal species of mangroves are shown in Figure 2.9.

Mangroves provide a safe haven for the offspring of many marine species; they also provide coastal protection by means of hydrodynamical damping of waves. Slopes of roughly 1:300 are possible, whereas mud flats have a slope of 1:1000 (Schiereck, 2000).

Unlike many other plants, mangroves are able to survive in salt water conditions. Seedlings are almost a complete tree when germinated, increasing the odds of survival in a dynamic situation and able to stand upright in a few days (Schiereck, 2000). However, the root system needs to be in contact with the open air regularly. This is partly an explanation for finding mangroves mostly in the upper tidal zone, but also makes mangroves susceptible for suffocation by means of excessive sedimentation (Schiereck, 2000). Susceptibility to excessive sedimentation does not imply ideal conditions for mangroves include no sedimentation at all: the most extensive areas of mangroves are found on sedimentary shorelines with natural sedimentation rates measured to be in the order of roughly $1 - 10\ mm/year$ (Ellison, 1999).

(a) *Avicennia* with "snorkel"-type roots. Image by Eunice Soh, retrieved from `http://naturallyours.blogspot.nl/2008/12/41a-st-johns-island.html`.

(b) *Rhizophora* with "prop"-roots or "stilt"-roots. Image by Bruce Holst/Marie Selby Botanical Gardens, retrieved from `http://eolspecies.lifedesks.org/node/1042`.

Figure 2.9: Mangrove species differ principally in the aerial root system.

2.5.2 Seagrass

Seagrasses are flowering plants, which get their name from their long, narrow leaves and the tendency to form large patches which somewhat resemble grassland. Seagrasses use photosynthesis to capture solar radiation. Water molecules are split in hydrogen and oxygen; hydrogen is then combined with carbon dioxide to form glucose and release oxygen.[1] Seagrass, like mangroves, also influence local water flow and provides shelter for marine species; nutrient recycling and food resources are also acknowledged traits (Orth et al., 2006). Seagrass is able to reproduce sexually or asexually; sexual reproduction results in the plant producing flowers and transferring pollen.[1] An example picture of seagrass can be found in Figure 2.10a.

2.5.3 Coral

Corals are animals often living in symbiosis with small algae called zooxanthellae, where the zooxanthellae utilize photosynthesis and can allocate energy towards the coral; corals also get their color from zooxanthellae. Corals get the majority of their food requirements from these zooxanthellae, but are also able to catch zoo-plankton with their tentacles. Corals are sessile (attached to substrate) and live in colonies of polyps; polyps are small, spineless animals (Larsen, 2012; Schutter, 2010, and sources cited within). "Scleractinian or reef-building corals are member of the phylum Cnidaria and are therefore closely related to other stinging animals such as jellyfish and sea anemones." (Schutter, 2010, p.10). These 'hard' corals can create calcium carbonate skeletons; over time these skeletons form the familiar sight of coral reefs.

Reproduction can occur either sexually or asexually, and within or outside coral. A common way of reproduction is spawning; the simultaneous release of eggs and sperm in the water. (Larsen, 2012, and sources cited within). An example picture of coral can be found Figure 2.10b. For a more detailed description of coral, see Section C.1.

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[1] http://www.seagrasswatch.org/seagrass.html

(a) Example of seagrass. Image taken by Eduardo Infantes Oanes, `http://ian.umces.edu/imagelibrary/displayimage-7220.html`.

(b) Example of (foliose) coral (Veron and Stafford-Smith, 2012).

Figure 2.10: Examples of seagrass and coral.

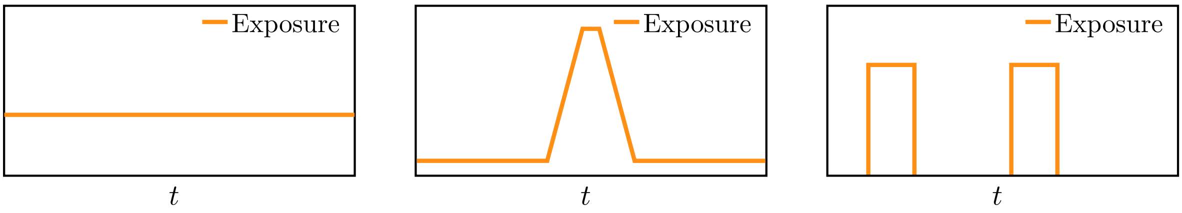
Chapter 3

Exposure assessment

After mentioning some simplified, general exposure patterns in Section 3.1, the first three phases in Figure 2.1 are treated consecutively in Section 3.2, Section 3.3 and Section 3.4; a short qualitative review on the type of exposures commonly connected to dredging and the uncertainty regarding the discussed exposure modeling can be found in Section 3.5.

3.1 Exposure characteristics

In general, a time record of an exposure such as TSS or turbidity shows variability. Recognizing patterns, or the absence of patterns, is an important part in analyzing and understanding measured exposures signals. Three basic forms which are often used are the constant, pulsed, and extreme event signal. Schematized illustrations of these three can be found in Figure 3.1.



(a) Constant exposure. (b) Peaked exposure. (c) Pulsed exposure.

Figure 3.1: Three simplified exposure patterns.

These three schematizations are known to be (over-)simplifications, yet are often used to characterize actual occurring exposure signals. The peaked exposure signal is often associated with extreme events, which result in rare (frequency wise), but high exposures. Pulsed exposures are often associated with recurring events such as a recurring seasonal combination of conditions; another example would be a tidal cycle.

3.2 Dredger, source of dredge plumes

Several dredging techniques are presently available to perform dredging operations, a few examples are shown in Figure 3.2. Besides the already shown Trailing Suction Hopper Dredger (TSHD) a number of other techniques are available, for example (Bray, 2008):

1. Cutter Suction Dredger (CSD)
2. Backhoe dredger
3. Grab dredger
4. Bucket ladder dredger
5. Settlement basin
6. Stationary suction dredger
7. New innovations like dustpan dredger, dipper dredger and bed-leveler

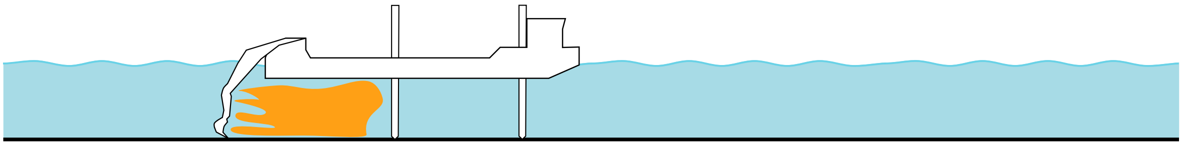
Each technique can result in a different contribution to a dredge plume: this is inherent to the type of dredging technique used and the associated production volumes per unit of time. Of particular interest in recent studies is the overflow phenomenon for hoppers, which is shown in Figure 3.2 for a TSHD. Overflow is the process of discharging excess water in order to reach a higher dredged material density in the hopper; almost inevitably the water is a mixture which also contains sediment. Globally, three phases can be distinguished as part of a loading cycle of a hopper dredge (Miedema and Vlasblom, 1995):

1. Filling the hopper with dredged material until the overflow level is reached.
2. Overflowing of the hopper in the overflow arrangement, while a part of the sediment settles in the hopper. How much of the sediment settles depends on the particular hopper overflow arrangement, sediment properties and grain size distribution.
3. Adjusting the overflow arrangement downwards to the sediment level, so the water above the sediment level can also be discharged. This way, the maximum hopper content or optimal loading is achieved.

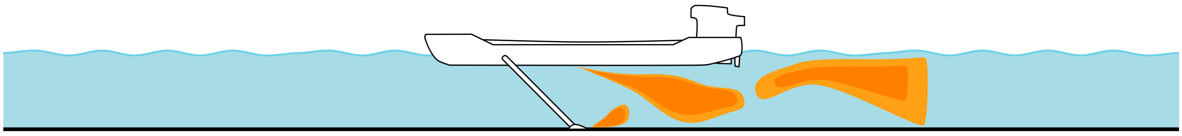
Estimating overflow source term

An example of the estimating of a source term is given here by describing overflow. Overflow is closely related to processes occurring during the filling and overflowing of the hopper. The duration of overflow (if any at all) is left out on purpose because this is largely a contractual/economical/practical optimization, even though this can have a significant influence on the contribution of overflow to a dredge plume. An example of a description of processes contributing to overflow is described in the TASS (Turbidity ASsessment Software) model (Spearman et al., 2011).

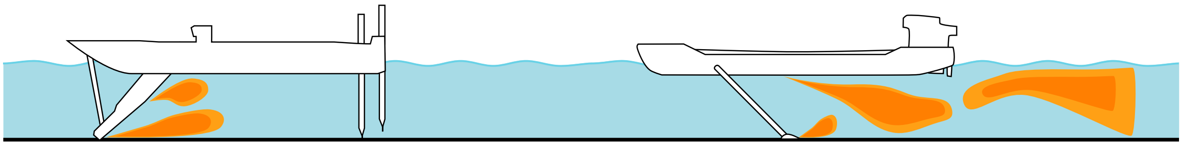
In this model, a hopper is conceptualized as being split in three zones: an input jet zone, a zone which covers most of the hopper where the primary effect of the input jet is negligible and an output zone/orifice where flow occurs through all of the hopper depth into the weir (Spearman et al. (2011); Figure 3.3a). The second zone is schematized in TASS by means of a 1dv model where both the horizontal advection and upward movement of water and sediment are modeled (Spearman et al. (2011); Figure 3.3b). Continuity is assumed here in a sense that horizontal discharge occurs in all layers along the hopper and out of the hopper because flow at the weir occurs in all directions (Spearman et al., 2011). By using a profile of velocities and sediment concentrations over the vertical instead of depth-averaged



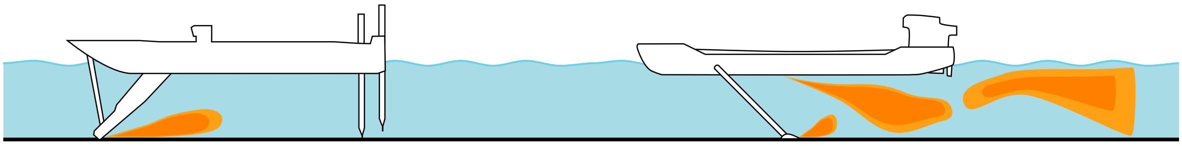
(a) Backhoe dredgers have relatively low production rates and the spill term is the bucket drip which is approximately evenly distributed over the water column.



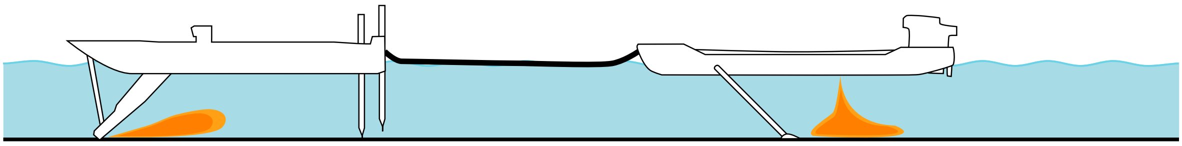
(b) Trailing Suction Hopper Dredger (TSHD) has spill terms at the draghead, overflow and behind the ship due to propeller wash re-suspension.



(c) Cutter Suction Dredger (CSD) combined with a TSHD. The CSD has spill terms at the cutterhead and behind the underwater pump. The TSHD collects the now loose material and has similar spill terms as a single TSHD.



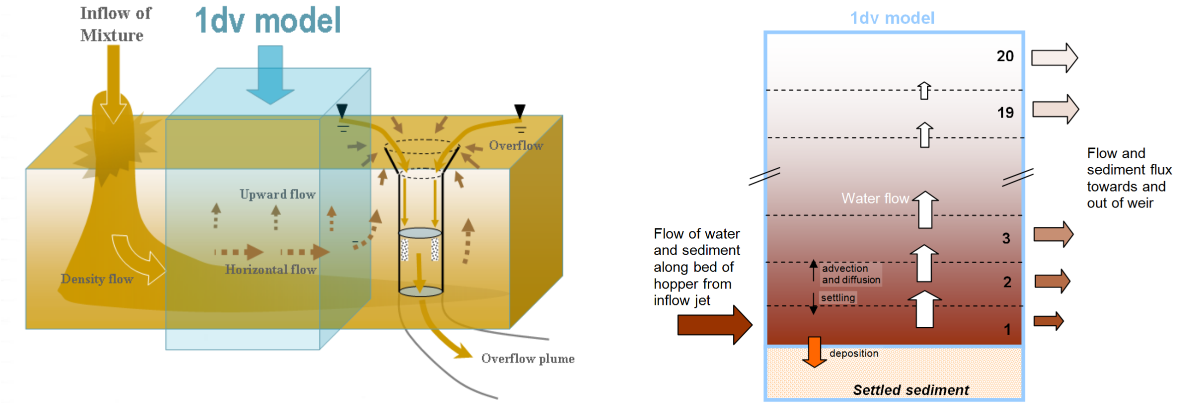
(d) Again a CSD combined with a TSHD, but now the CSD only crushes - no pumping (higher RPM/thinner layers). This removes the spill term behind the underwater pump.



(e) CSD linked by means of a floating pipeline to a TSHD. This method removes the CSD spill term behind the underwater pump (because it is pumped all the way to the TSHD). It also removes the TSHD spill terms at the draghead and due to propeller-wash: the draghead is not used, and the TSHD is considered to be stationary.

Figure 3.2: Popular dredging methods using various dredging equipment. For the TSHD, overflow is assumed in all cases, though it could be a consideration to disallow overflow: this ultimately results in more trips to the dump site. The size of drawn spill terms is purely illustrative and cannot be used to compare spill terms. *Taken from personal communication with Mark van Koningsveld (Van Oord Dredging and Marine Contractors, P.O. Box 8574, 3009 AN, Rotterdam, The Netherlands)*

velocities and concentrations improves accuracy of the settling flux and the slow increase in concentration that is often observed in dredging overflow (Spearman et al., 2011). This schematization results in the clearer layers of water near the surface do not dominate the sediment output in the overflow (Spearman et al., 2011).



(a) Conceptualization of the hopper as presented in TASS (Spearman et al., 2011).

(b) Schematisation used in the 1dv model in TASS (Spearman et al., 2011).

Figure 3.3: Important concepts in Turbidity Assessment Software (TASS) as proposed in Spearman et al. (2011) for schematization of the overflow term.

3.3 Dynamic phase of a dredge plume

While the previous section gave a brief description of source terms of a dredge plume (first phase in Figure 2.1), the second and third phase describe the actual development of the dredge plume. The development of a dredge plume can be divided in a dynamic and a passive phase, corresponding to the second and third phase in Figure 2.1.

According to Lee and Chu (2003), “Plumes are fluid motions that are produced by continuous sources of buoyancy”. In the case of overflow discharges of a water-sediment mixture form a TSHD, a negative-buoyant, or dynamic, plume will be formed (Dankers, 2002; Becker, 2011). A part of this plume will hit the bottom and will propagate as a density current, while other parts will mix with the surrounding water and form a passive plume. Dominant processes in the passive plume are diffusion and advection which are part of the hydrodynamic environment. Other factors are the currents causing advection and turbulence, waves, wind speed and wind direction. For both the dynamic and passive phase the amount of sediment, grain size and distribution, disaggregation properties and the amount of energy put into the dredging operation will affect the plume (Becker, 2011).

The dynamic phase of a dredge plume is particularly hard to understand and quantify due to the presence of quite some interacting and complicated processes: cross flow due to currents, propeller mixing, and hull flow are just a few of these processes (De Wit, 2010). However, if a quantification can be made for the dynamic phase as a ‘source term for the passive phase’, further development of the passive phase is relatively straightforward because advection and diffusion are well understood processes. This can be modeled with an hydrodynamic (and transport) simulation program, such as Delft3D or MIKE 21/3. While this indicates a strict separation between the dynamic and passive phase, in reality a transitional zone exists between the dynamic and passive phase.

Source term for passive phase

Instead of working with source terms which correspond directly to the source terms of a dredger, in practice these source terms are often translated to source terms for the passive phase, skipping processes of the dynamic phase. By essentially skipping the (difficult) modeling of complex interacting processes such as hull flow and cross flow, the influence of the dynamic phase is captured in a conservative estimate, based on practical experience, for a source term for the passive phase. But because of the conservative nature of these estimates, actual numbers can be considerably lower. In John et al. (2000), four ways were given to represent a source term for a dredge plume (Becker, 2011):

1. Sediment concentration increases in the vicinity of the dredging activity (mg/L)
2. Rate of release of sediment into the water column per unit of time (kg/s)
3. 'S'-factor approach, in which the total mass of sediment put into suspension is expressed relative to the quantity of material that is dredged (kg/m^3)
4. Sediment flux method, which describes the sediment loss through the boundaries of a designated area within which the dredger is working

Method 1 is considered to be site specific and and therefore not universally applicable (Van Eekelen, 2007; Becker, 2011). The second method is already used often in plume modeling and the proposed release rate be readily used as a source term: though material type and near-field/dynamic phase plume behavior still need to be included (Whiteside et al., 1995; Becker, 2011). Method 3, proposed by Pennekamp et al. (1996), introduces an S-factor which is dependent on soil class, dredger type ambient conditions and planned use of the chosen equipment. Usable as a first estimate, this method has the advantage of re-usable factors (Becker, 2011). The last method has been used Øresund link project but is more a measuring method (as opposed to a resuspension description); extensive measuring is necessary (Becker, 2011). A combination of method 2 and 3 was considered in Becker (2011) as appropriate, in order to combine the soil and dredging equipment characteristics of method 3 with the time dependent release of method 2. The magnitude of the source terms for each type of equipment is based on project specific information about the dredged materials and the expected amount of fines in it. An estimate of the total mass of fines $(m_{fines;dry})$ is made in equation [3.1] by multiplying the dredged volume $(V_{dredged})$, dry mass of dredged material $(\rho_{dredged;dry})$ and the fraction of fines $(frac_{fines})$. The fraction of fines which actually enters the passive phase, both at the dredge site and dump site, depends on the work method.

$$\begin{aligned} m_{fines;dry} &= V_{dredged} \times \rho_{dredged;dry} \times frac_{fines} \\ m_{fines;dry;passive;total} &= m_{fines;dry} \times (Dredge + Dump) \\ m_{fines;dry;passive;dredge} &= m_{fines;dry} \times Dredge \\ m_{fines;dry;passive;dump} &= m_{fines;dry} \times Dump \end{aligned} \qquad [3.1]$$

Different work methods have different influencing factors. Three popular work methods are captured with their influencing factors in equations [3.2], [3.3] and [3.4]. In this equation, a number of variables are introduced. For the TSHD, there are three spill terms attributing to a passive plume: dragheads ($\%_{drag}$), propeller wash($\%_{prop}$) and overflow ($\%_{over}$). The contribution of overflow is modified by the overflow ratio (r_{over}) and the trapping factor ($\%_{trap}$). If there is no overflow the overflow ratio equals zero, leading to no contribution of fines to the passive plume from the overflow. The propeller wash term is especially sensitive for limited keel clearance. BHD and CSD both are considered to only have a single spill term: this is bucket drip ($\%_{bucket}$) and cutterhead spill ($\%_{head}$) respectively. The percentage actually lost at dump sites depends quite a lot on the type of dumping, e.g. if a settlement basin is used. If the latter is true, a significant reduction or even negligible contributions to a passive plume can be expected - settlement basin calculations can provide more accurate numbers if necessary.

$$CSD \left\{ \begin{aligned} Dredge &= \overbrace{\%_{head}}^{\textit{Cutterhead}} \\ Dump &= \overbrace{\%_{dump}}^{\textit{Dumping}} \times \overbrace{(1 - \%_{cutterhead})}^{\textit{Total fraction of fines NOT lost at dredge site}} \end{aligned} \right. \tag*{[3.2]}$$

$$BHD \left\{ \begin{aligned} Dredge &= \overbrace{\%_{bucket}}^{\textit{Bucket drip}} \\ Dump &= \overbrace{\%_{dump}}^{\textit{Dumping}} \times \overbrace{(1 - \%_{bucketdrip})}^{\textit{Total fraction of fines NOT lost at dredge site}} \end{aligned} \right. \tag*{[3.3]}$$

$$TSHD \left\{ \begin{aligned} \%_{d;p} &= \overbrace{\%_{drag}}^{\textit{drag head}} + \overbrace{\%_{prop}}^{\textit{propeller}} \\ Dredge &= \%_{d;p} + \overbrace{r_{over} \times (1 - \%_{d;p}) \times (1 - \%_{trap})}^{\textit{Total fraction of fines overflowed}} \times \overbrace{\%_{over}}^{\textit{overflow}} \\ Dump &= \overbrace{\%_{dump}}^{\textit{Dumping}} \times \overbrace{(1 - (\%_{d;p} + r_{over} \times (1 - \%_{d;p}) \times (1 - \%_{trap})))}^{\textit{Total fraction of fines NOT lost at dredge site}} \end{aligned} \right. \tag*{[3.4]}$$

Equations and descriptions taken from personal communication with Mark van Koningsveld (Van Oord Dredging and Marine Contractors, P.O. Box 8574, 3009 AN, Rotterdam, The Netherlands).

In general equations [3.2], [3.3] and [3.4] show that, for the same mass of fines, somehow reducing the amount of fines spilled at the dredge location (e.g. no overflow) equals more fines being released at the dumping location. The type of dumping strategy (e.g. settlement basin) has a considerable influence on the amount of fines attributing to a passive plume near the dumping site. Restricting or disallowing overflow equals more trips being made resulting in higher fuel usage and, in the case of a TSHD, also includes a higher number of lowering/raising the dragheads: this could lead to in increased risk of impact on marine life.

3.4 Passive phase of a dredge plume

Phase three in Figure 2.1, the passive phase, is better understood than the dynamic phase of a dredge plume. In this phase, the dominant processes are assumed to be advection and diffusion which are processed in ambient water. The passive plume responds to these processes; it responds to the hydrodynamic environment. Other influences include the (fine) sediment material properties, currents (causing advection and turbulence), wind and waves (Becker, 2011). A common assumption is rapid mixing of the surrounding water and the passive plume; resulting in depth averaged material concentrations (Becker, 2011).

In order to quantify the passive plume at a location of interest, using the source terms of Section 3.3, a hydrodynamic (and transport) simulation program is needed; this can be done by using programs such as the open source hydrodynamic model Delft3D, or the hydrodynamic part of MIKE software (MIKE 21 HD or MIKE 3 HD). In the following Delft3D is used to give an indication of the inner workings of such a simulation program, but the governing equations should be largely applicable to MIKE software as well; though different choices can be made for numerical approximation and boundary formulation and approximation.

Please note: *In this section (Section 3.4) a direct citation of Becker (2011) is given regarding the conceptual description of passive plume modeling (specifically the sections regarding flow, sediment transport modeling and DelftDashboard); for more details either Chapter 3 of Becker (2011) or the the manuals of* Delft3D *and/or* MIKE *software can be used. A direct citation with only slight modifications is chosen here because the focus of the report lies, as mentioned in Section 1.2.1, on the effect schematization. But, insight in schematizations on the exposure side are indispensable for a complete risk assessment. Quotation marks have been left out for readability and brevity.*

A suitable candidate for including the source term in a model is the Dredge Plume Toolbox of the pre-processing tool DelftDashboard, which is part of OpenEarth (Van Koningsveld et al., 2010). DelftDashboard is a (standalone) user interface written in Matlab with a large number of (coupled) toolboxes which should increase the speed of setting up a model (Becker, 2011). It is currently coupled to Delft3D for doing hydrodynamic and transport calculations, but because the interface is (crudely put) a wrapper around Delft3D, it should be relatively straightforward to make it use MIKE software as well.

In the case of Delft3D flow, a model is set up which calculates non-steady flow and transport phenomena resulting from meteorological and tidal forcing: these can either be solved in two (depth averaged) or three dimensions. Governing equations are the horizontal equations of motion, continuity equation and transport equations for conservative constituents. Flow is forced at the open boundaries by tide; wind provides stress at the free surface and pressure gradient due to (barotropic) free surface gradients or (baroclinic) density gradients (Becker (2011) and sources cited within).

3.4.1 Flow modeling

The 2DV non-linear shallow water equations are derived from the three dimensional Navier Stokes equations for incompressible free surface flow. Several assumptions and approximations are used, the main three being the Boussinesq approximation, the eddy viscosity concept and the assumption of shallow water. The set of partial differential equations in combination with an appropriate set of initial and boundary conditions is solved on a finite difference grid (Becker, 2011).

The depth-averaged continuity equation is given by (Becker, 2011):

$$\frac{\partial \zeta}{\partial t} + \frac{\partial HU}{\partial x} + \frac{\partial HV}{\partial y} = 0, \tag{3.5}$$

where $H = \zeta + d$, with ζ the water level and d the depth and U and V are depth-averaged velocities in x- and y-direction respectively. The depth-averaged momentum equations in x- and y-direction are given by (Becker, 2011):

$$\frac{\partial U}{\partial t} + U\frac{\partial U}{\partial x} + V\frac{\partial U}{\partial y} = fV - g\frac{\partial \zeta}{\partial x} - \frac{1}{\rho_0}P_x + \frac{1}{\rho_0 H}(\tau_{sx} - \tau_{bx}) - \nu_H \nabla^2 U, \tag{3.6}$$

$$\frac{\partial V}{\partial t} + U\frac{\partial V}{\partial x} + V\frac{\partial V}{\partial y} = -fU - g\frac{\partial \zeta}{\partial y} - \frac{1}{\rho_0}P_y + \frac{1}{\rho_0 H}(\tau_{sy} - \tau_{by}) - \nu_H \nabla^2 V, \tag{3.7}$$

where f is the Coriolis parameter, g is the acceleration due to gravity, ρ_0 is the reference density of water, P_x and P_y represent the pressure gradients, τ_{sx} and τ_{sy} are the surface wind stress in the x- and y-direction, τ_{bx} and τ_{by} are bottom frictional stress in the x- and y-direction and ν_H is the horizontal eddy viscosity coefficient. The Laplace operator is defined as $\nabla^2 = \frac{\partial^2}{\partial x^2} + \frac{\partial^2}{\partial y^2}$. Equation 3.5, Equation 3.6 and Equation 3.7 represent the 2D depth-averaged shallow water equations (Becker, 2011). For 2D depth-averaged flow the shear-stress at the bed induced by a turbulent flow is assumed to be given by a quadratic friction law (Becker, 2011):

$$\vec{\tau}_b = \frac{\rho_0 g \vec{U}|\vec{U}|}{C_{2D}^2}, \tag{3.8}$$

where $|\vec{U}|$ is the magnitude of the depth-averaged horizontal velocity and C_{2D} is the 2D-Chézy coefficient (Becker, 2011). At the free surface, the boundary conditions for the momentum equations require a formulation for the surface stress. Without wind, the stress is zero. The magnitude of the wind shear-stress is determined by (Becker, 2011):

$$|\vec{\tau}_s| = \rho_a C_d U_{10}^2, \tag{3.9}$$

where ρ_a is the density of air, U_{10} is the wind speed 10 meter above the free surface (time and space dependent) and C_d is the wind drag coefficient, dependent on U_{10}. At the open water boundaries, data needed for the boundary conditions can be obtained from measurements, tide tables or from a larger model, which encloses the model at hand (nesting) (Becker, 2011).

3.4.2 Sediment transport modeling

Sediment transport and morphology are supported in Delft3D-FLOW; bed-load and suspended load transport of non-cohesive sediments and suspended load of cohesive sediments can be modeled. The advection-diffusion (mass-balance) equation that has to be solved for 2D, depth-averaged transport of suspended sediment is given by (Becker, 2011):

$$\frac{\partial HC^{(\ell)}}{\partial t}+\frac{\partial HUC^{(\ell)}}{\partial x}+\frac{\partial HVC^{(\ell)}}{\partial y}- \frac{\partial}{\partial x}\left(\varepsilon_{s,x}^{(\ell)}\frac{\partial HC^{(\ell)}}{\partial x}\right)-\frac{\partial}{\partial y}\left(\varepsilon_{s,y}^{(\ell)}\frac{\partial HC^{(\ell)}}{\partial y}\right)-\left(D^{(\ell)}-E^{(\ell)}\right)=0, \quad [3.10]$$

where $H=\zeta+d$, with ζ the water level and d the depth, $C^{(\ell)}$ is the depth-averaged mass concentration of sediment fraction (ℓ), U and V are depth-averaged flow velocity components, $\varepsilon_{s,x}^{(\ell)}$ and $\varepsilon_{s,y}^{(\ell)}$ are eddy diffusivities and $D^{(\ell)}$ and $E^{(\ell)}$ are the rates of sediment deposition and erosion respectively (Becker, 2011). Sediment is different from ordinary constituents, such as salinity and heat, since it is exchanged between the bed and the flow and it settles due to the action of gravity. The settling velocity, deposition and erosion are processes that are sediment-type specific. The exchange of suspended sediment is determined by the flux from the bed to the bottom layer and vice versa. In every cell a source and sink term are then applied and the bed level is updated (Becker, 2011). To calculate the exchange between water phase and bed, the Partheniades-Krone formulations are used for erosion and deposition (Partheniades, 1965; Becker, 2011):

$$E^{(\ell)}=M^{(\ell)}S(\tau_{cw},\tau_{cr,e}^{(\ell)}), \quad [3.11]$$

$$D^{(\ell)}=w_s^{(\ell)}C^{(\ell)}S(\tau_{cw},\tau_{cr,d}^{(\ell)}), \quad [3.12]$$

where $M^{(\ell)}$ is the erosion parameter, $S(\tau_{cw},\tau_{cr,e}^{(\ell)})$ is an erosion step function, $w_s^{(\ell)}$ is the (hindered) sediment settling velocity of sediment fraction (ℓ), $S(\tau_{cw},\tau_{cr,d}^{(\ell)})$ is a deposition step function, τ_{cw} is the maximum bed shear stress, $\tau_{cr,e}^{(\ell)}$ is the critical erosion shear stress and $\tau_{cr,d}^{(\ell)}$ is the critical deposition shear stress (Becker, 2011). Cohesive sediment tends to form flocs when it is suspended in salt water. These flocs are larger than the particles they consist of and have a higher settling velocity. For single mud flocs with a fractal structure in still water, a formula for the settling velocity can be obtained from a balance between gravitational and drag force. For spherical, Euclidean particles in the Stokes' regime, where $Re_f \ll 1$, the Stokes' formula for a stationary settling particle reads (Van Rijn, 1984; Winterwerp and van Kesteren, 2004; Becker, 2011):

$$w_{s,r}=\frac{(\rho_s-\rho_w)gD_f^2}{18\mu}, \quad [3.13]$$

where D_f is the representative mud floc diameter and μ is the dynamic viscosity. To take into account flocculation effects and hindered settling, Van Rijn (2007) proposes the following equation for the sediment settling velocity (Becker, 2011):

$$w_s=\phi_{floc}\phi_{hs}w_{s,r}, \quad [3.14]$$

Symbol	Description
x^i_j	x co-ordinate of waypoint j of dredge track i
y^i_j	y co-ordinate of waypoint j of dredge track i
t^i_{start}	Start time of dredge track i
t^i_{stop}	Stop time of dredge track i
Q^i_{start}	Discharge at t^i_{start} for dredge track i
Q^i_{stop}	Discharge at t^i_{stop} for dredge track i
c^i_{start}	Concentration at t^i_{start} for dredge track i
c^i_{stop}	Concentration at t^i_{stop} for dredge track i
$t^n_{c;start}$	Start time of dredge cycle n
d^n_c	Duration of dredge cycle n
N_c	Amount of simulated dredge cycles

Table 3.1: Parameters that are applied in the Dredge Plume toolbox (Becker, 2011).

where ϕ_{floc} is the flocculation factor and ϕ_{hs} is the hindered settling factor. For a salinity $Sa \geq 5ppt$ and particles finer than $D_{sand} = 63\ \mu m$, the flocculation factor is given by (Becker, 2011):

$$\phi_{floc} = [4 + \log_{10}(2c/c_{gel})]^{\alpha},$$
$$\text{with a minimum value of 1 and a maximum value of 10,} \quad [3.15]$$

where $\alpha = (D_{sand}/D_{50}) - 1$, with $\alpha_{min} = 0$ and $\alpha_{max} = 3$; c is the mass concentration $(= \rho_s c_{volume})$ and c_{gel} is the gelling mass concentration (between 130 and 1722 kg/m^3). Hindered settling is negligible due to the low suspended sediment concentrations in dredge plumes (i.e. $\phi_{hs} = 1$) (Becker, 2011).

3.4.3 Source term using DelftDashboard

The Dredge Plume toolbox consist of a series of MATLAB routines which translate a user defined dredge track into a series of discharge locations (m, n) and corresponding start and stop times. The source term is represented by a discharge Q and a concentration c. The input parameters are depicted in Table 3.1. The values in between Q_{start} and Q_{stop}, and c_{start} and c_{stop} are obtained by linear interpolation. This allows for a non-steady source term, representing the hopper loading curve (e.g. Miedema and Vlasblom, 1995) (Becker, 2011).

Important limitations occur when using this toolbox, as mentioned in (Becker, 2011):

- For reasons of computational efficiency a two dimensional (2D, depth-averaged) grid is imposed. To be able to do this, the fluid has to be vertically homogeneous. For far-field suspended sediment plume simulation, the flow regime can assumed to be vertically well-mixed, which makes the simplification a reasonable choice.
- Current iterations of DelftDashboard only allow a rectilinear grid; this makes modeling rivers, estuaries or any other curved boundaries crudely modeled which can lead to significant errors.

- Negative sediment concentrations can be computed; these can be filtered out but the filtering requires a significant increase in computation time.
- Hindered settling is not part of the schematization; though this is expected to have a significant influence for high concentrations (opposite to what is normally expected for a passive plume). Flocculation is part of the calculation as it can have a significant effect on particle fall velocity. However, interaction between different sediment types leading to possible flocculation dynamics are not modeled.
- Wave action is left out in the schematization of (Becker, 2011); it would very much depend on the local situation if this is a reasonable simplification or not. If wave action is expected to add significant additional diffusion (which will mostly be near the free surface), it should be included.
- As mentioned before, in Becker (2011) the dynamic phase is skipped by directly using source terms adjusted to expected influences of the dynamic phase. This means the dynamic phase is not modeled explicitly.

3.5 Dredging exposures and variability

3.5.1 Exposures related to dredging

In essence, dredging causes sediment to be brought into suspension in the water column which has been described using the description of source → dynamic phase → passive phase in Section 3.2, Section 3.3, and Section 3.4. At first, a wide range of particle sizes will be present. However, often the larger particles will settle relatively close to the dredging operation or dump site (depending on the work method, sediment characteristics/composition and dynamic phase processes). The finer particles take much longer to settle and consequently can travel a much larger distance: These fine particles form the aforementioned described (passive) dredge plume. The physical parameters for assessing exposure due to dredge plumes which are reviewed often in practice (in relation to environmental assessments) are sedimentation and suspended (fine) sediment; the latter is usually expressed in turbidity and/or Total Suspended Solids (TSS). These parameters are related, but certainly not the same:

1. Suspended solids comprise fine particles of inorganic solids (e.g. clay, silt and sand) and organic solids (e.g. algae and detritus) suspended in the water column. Suspended solids are usually measured as Total Suspended Solids (TSS), the dry weight of suspended solids per unit volume of water. TSS is usually reported in milligrams of solids per liter of water (mg/L) (Bray, 2008).
2. Turbidity is an optical property of water that causes light to be scattered and absorbed rather than transmitted in straight lines through the water. It is caused by the molecules of the water itself, dissolved substances and organic and inorganic suspended matter. Turbidity is measured in various optical units such as Secchi depth, Jackson Turbidity Units (JTU), and Nephelometric Turbidity Units (NTU) (Bray, 2008; Thackston and Palermo, 2000). Turbidity is inversely related to transparency of the water.

The relation between suspended particles and turbidity can be found in the ability of suspended particles to scatter light; size, shape and relative refractive index of the particles.

However the wavelength of the light also has a role. For example, this could result in two equal samples with equal TSS, but rather different turbidity. To further complicate matters, the turbidity measuring equipment can also produce significant different turbidity readings. Worse, even at the same site the relationship between turbidity and TSS does not have to be constant. All this indicates that great care has to be taken with any relationship describing a transformation from TSS to turbidity, or reversed. If turbidity or TSS needs to be modeled or forecast, the preferred method is not using any transformation relationship at all, but only use TSS or turbidity data originating from respectively TSS or turbidity measuring equipment which has been calibrated to local circumstances (Bray, 2008; Thackston and Palermo, 2000).

In addition, turbidity does not correlate well with many categories of environmental effect, while TSS can correlate reasonably well with environmental effects. The reason why turbidity measurements are still used is because turbidity can be measured quickly, while TSS takes considerably longer to quantify (roughly ranging from a few hours to a day) (Thackston and Palermo, 2000). Unfortunately, practical limitations often require the transformation of turbidity into TSS (or the other way around).

Settling of (assumed to be predominantly fine) sediment can also be a significant exposure. An often made assumption is, concerning dredge plumes, choosing a location to be inspected closer to a dredging or dumping location will result in more and coarser sediment to settle. In general, moving further away from the dredge or dump location it is expected that more and more fine sediments will be dispersed, spread or settled: this will alleviate dredging induced TSS/turbidity and settlement exposures. This will continue to a point where dredging induced exposures cannot be found to significantly alter background exposures; even though significant exposures of dredge plumes can be found several kilometers away from the dredging or dumping location.

What has not been mentioned yet, but can be a very significant part in assessing possible exposures due to dredging, is the release of previously bound contaminants, or other factors which can induce significant changes to water quality. It is acknowledged that these factors are important, but are not part of this thesis.

3.5.2 Uncertainties related to exposure modeling

A significant amount of transformations have been made in order to translate source terms from a dredger to passive plume exposures at a location of interest; inherently this means adding uncertainty to the final answer. A number of important uncertainties in the translation of source term to passive plume exposures at a location of interest are:

1. Inherent and/or epistemic uncertainty in sediment type, distribution, size and characteristics.

2. Inherent and/or epistemic uncertainty in the source terms of a dredger.

3. Inherent and/or epistemic uncertainty in the transformation of source term of a dredger to source term as input for a passive plume

4. Inherent and/or epistemic uncertainty in the (predicted) weather conditions (for example wind speed and wind direction)

5. Inherent and/or epistemic uncertainty in the location of interest
6. Inherent and/or epistemic uncertainty due to model limitations, approximations and simplifications as mentioned in Subsection 3.4.3

Item three of the list is mentioned because the transformation of source terms is usually based on conservative experience-based estimates. In order to properly model these in a probabilistic calculation, the deterministic values for the source terms should not be taken as average values, but (for example) as a 95% upper limit. Distribution type and parameters can be derived by comparing measured values with predicted values, or using more experience based numbers.

Item four and implications regarding probabilistic calculations are also mentioned in Becker (2011): two methods are used to approach this problem. The first implements a Monte Carlo approach where weather conditions are modeled as stochastic variables, and a large number of hydrodynamic/sediment transport calculations are made each with a new set of randomly drawn parameters from the predefined stochastic variables. This means a lot of computational time is needed, or the model needs to be heavily simplified by sacrificing detail in order to cut on computational time (e.g. increasing grid size). A second method would still include setting up weather conditions as stochastic variables, but predetermine a limited number of 'normative situations' which can still capture a significant portion of the variability of the final answer, but don't require as much computational time.

The fifth point might seem odd, but is mentioned here to emphasize the often made simplification of an area of interest to a point of interest: if the area of interest is large this simplification won't as the exposures by a dredge plume might differ significantly spatially for the area of interest, and the area of interest might have a gradient of characteristics (e.g. in the case of a coral reef, distribution of coral species can vary spatially; each species can have significant different traits).

Chapter 4

Effect Assessment

Effects are looked at from different angles: characteristics (or traits) of a sensitive receiver in Section 4.1, characteristics of of different relevant (background) exposures in Section 4.2 and characteristics of the relation between exposure and effect in Section 4.3. Unifying these angles in a single approach is looked at in Section 4.4.

4.1 Receiver characteristics

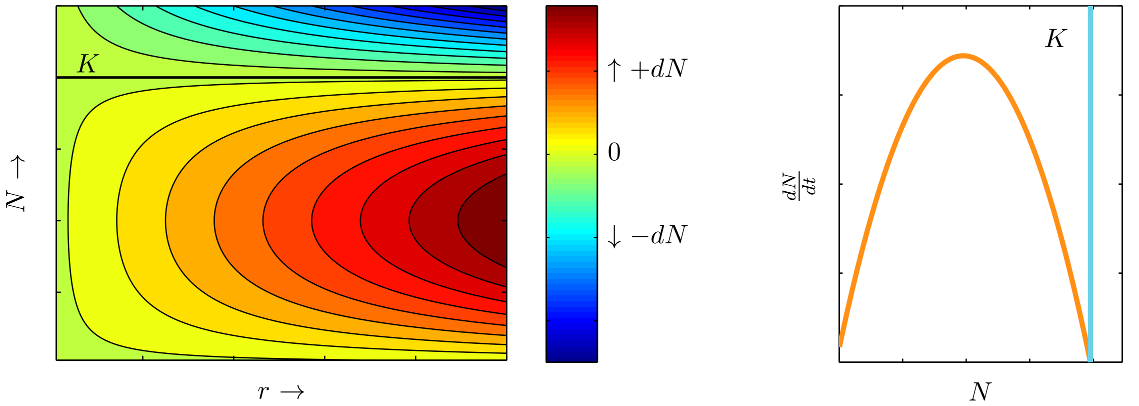
Exposures (Chapter 3) lead to effects; but the type, number and extensiveness of effects are determined not only by the conditions and resources (as mentioned in Subsection 2.4.2), but also by the characteristics of receivers: even individuals from the same species can have different characteristics because of small differences between individuals.

4.1.1 Life history theory

Not only individuals exhibit differences, this is also true for populations/species and communities. The life history theory states that the duration and schedule of events for an organism are based on 'fitness', in order to have the largest number of surviving offspring. A simplified explanation of the life history theory can be explained by r/K selection theory. Communities can be divided into the generalized categories of r-strategists and K-strategists. The name of these categories are drawn from a standard model of population dynamics (equation [4.1], by Verhulst). In this equation N is the population number, K is the carrying capacity of the local environment and r is the maximum growth rate. Equation [4.1] has also been plotted in Figure 4.1.

$$\frac{dN}{dt} = rN\left(1 - \frac{N}{K}\right) \qquad [4.1]$$

K-strategists (high K, low r, "equilibrium") can typically be found in a habitat that is calm (calm defined here as not having too many large unsystematic variations), which results in fierce competition between offspring and/or adults: this leads respectively to fewer, larger offspring or in a smaller reproductive allocation and larger size adults. On the other hand, r-strategists (low K, high r, "opportunistic") are generally associated with much more dynamic situations. These conditions can lead to more, smaller offspring or a relative large investment



(a) Values of dN for a constant K and a range of strictly positive values for N and r.

(b) dN/dt for constant r and N smaller than K.

Figure 4.1: Fictitious examples belonging to equation [4.1].

in reproductive abilities resulting in smaller adults (Begon et al., 1990). Figure 4.1b is a sample from Figure 4.1a and shows that, for population sizes N below capacity K and constant r, the rate of change in population size dN is not constant: the rate of change is largest at intermediate densities, while both low and high densities have lower growth rates. Lower reproductive capability at low densities and greater competition at high densities explain these lower growth rates. Typically new habitats are first colonized by r-strategists, but when densities increase, the portion of K-strategists increases at the expense of r-strategists. Differences in speed of recovery after significant damage might therefore be explained by the proportion K-strategists versus r-strategists and the relative density of the community.

Application to seagrass

The simplification made in the r/K selection theory in comparison to the more general life history theory is the assumption of two discrete classes ('fast' r or 'slow' K), while in reality a gradient would be more likely. This is shown for a number of seagrass species in a conceptual diagram in Figure 4.2. While the gradient between 'fast' r and 'slow' K adds flexibility to the r/K classification, a discrete number of groups is still easier to interpret; questionable remains how many groups can (or should) be discerned. A third group 'beyond K' has been proposed which is stress-adapted and can survive in unfavorable conditions (Darling et al., 2012, and references therein). However, there is no reason to assume three is the definitive amount of groups applicable.

Application to coral

In Darling et al. (2012), grouping of 143 reef coral species was done based on 11 traits. Emphasis is placed on the fact that these are species traits, not responses to different conditions. The 11 traits used in Darling et al. (2012) are colony growth form, solitary or colonial colonies, reproductive mode, colony size, skeleton diameter, depth range, fecundity (number of eggs per individual), generation length (years between generations), average growth rate, skeletal density of $CaCO_3$ and lastly symbiont richness. Best fits were achieved for models with two, three or four clusters (Darling et al., 2012). For a more general description of corals and terms

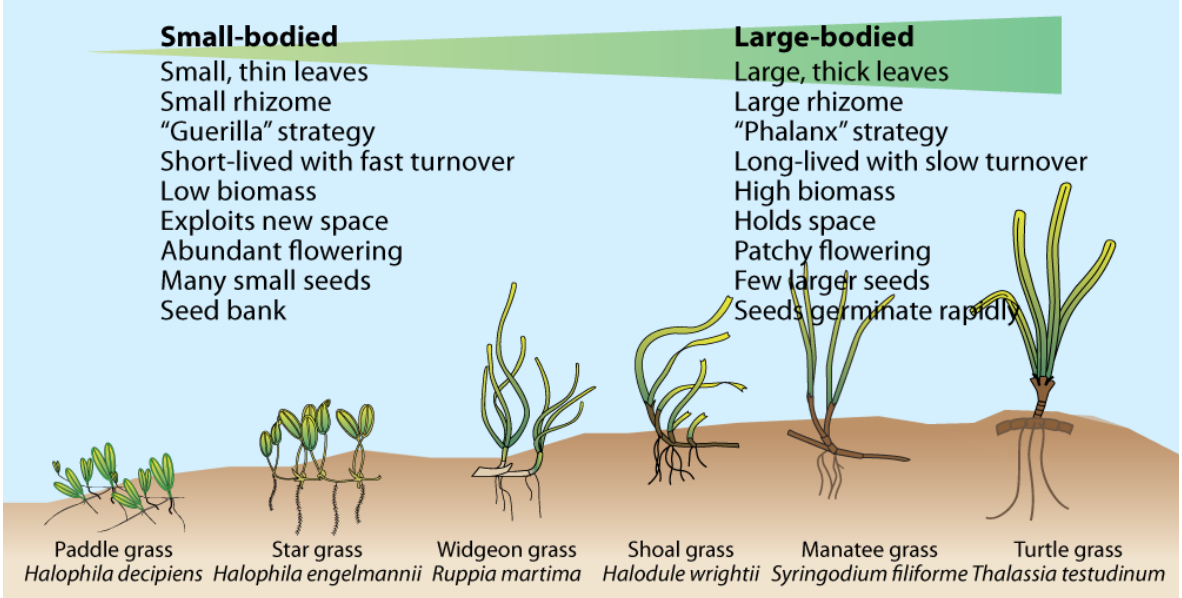


Figure 4.2: A gradient of r to K strategy for seagrass. Image by Kruczynski, W.L., and P.J Fletcher (eds.). 2012. Tropical Connections: South Florida's marine environment.

associated with corals, see Appendix C. Three of the four clusters coincide with the three groups in the C-S-R triangle, which is an application to plants of the Universal Adaptive Strategy Theory hypothesized by J. Philip Grime (University of Sheffield, UK) and Simon Pierce (University of Milan, Italy); the three groups distinguished in C-S-R are Competitor, Stress tolerator and Ruderal.

The four cluster model is shown in Figure 4.3. The first group could be said to be 'competitive': efficient at using resources, dominant in productive environments but sensitive to storm damage and significant temperature changes (Darling et al., 2012, and references therein). Crudely put, species in this group only thrive under ideal conditions. This group consists of large, branching and plating species that grow quickly, occur at shallow depth and reproduce by broadcast spawning (Darling et al., 2012).

The second group would represent a ruderal (or 'weedy') strategy: they reproduce faster and can opportunistically colonize recently disturbed habitats (Darling et al., 2012, and references therein). Reproduction is done by brooding and colony sizes are generally smaller.

The third group is called a generalists group in Darling et al. (2012) which can be found in habitats where competition is limited by low levels of stress (Darling et al., 2012). Alternatively, the group may be labeled as somewhat more competitive than the stress-tolerant fourth group. However, specific traits characteristic of this group were not found, and species in this group need a closer look if they actually are a separate group or part of any of the other three groups (Darling et al., 2012).

The fourth group would represent a stress-tolerant group. The group consists of slow growing, broadcast spawning corals which are 'dome' shaped (massive, submassive and encrusting growth forms). Traits which are considered beneficial for members of this group are longer generation times, large skeletal sizes (energy storage), high fecundity and persistence in case recruitment fails (even for sustained failure) (Darling et al., 2012, and references therein).

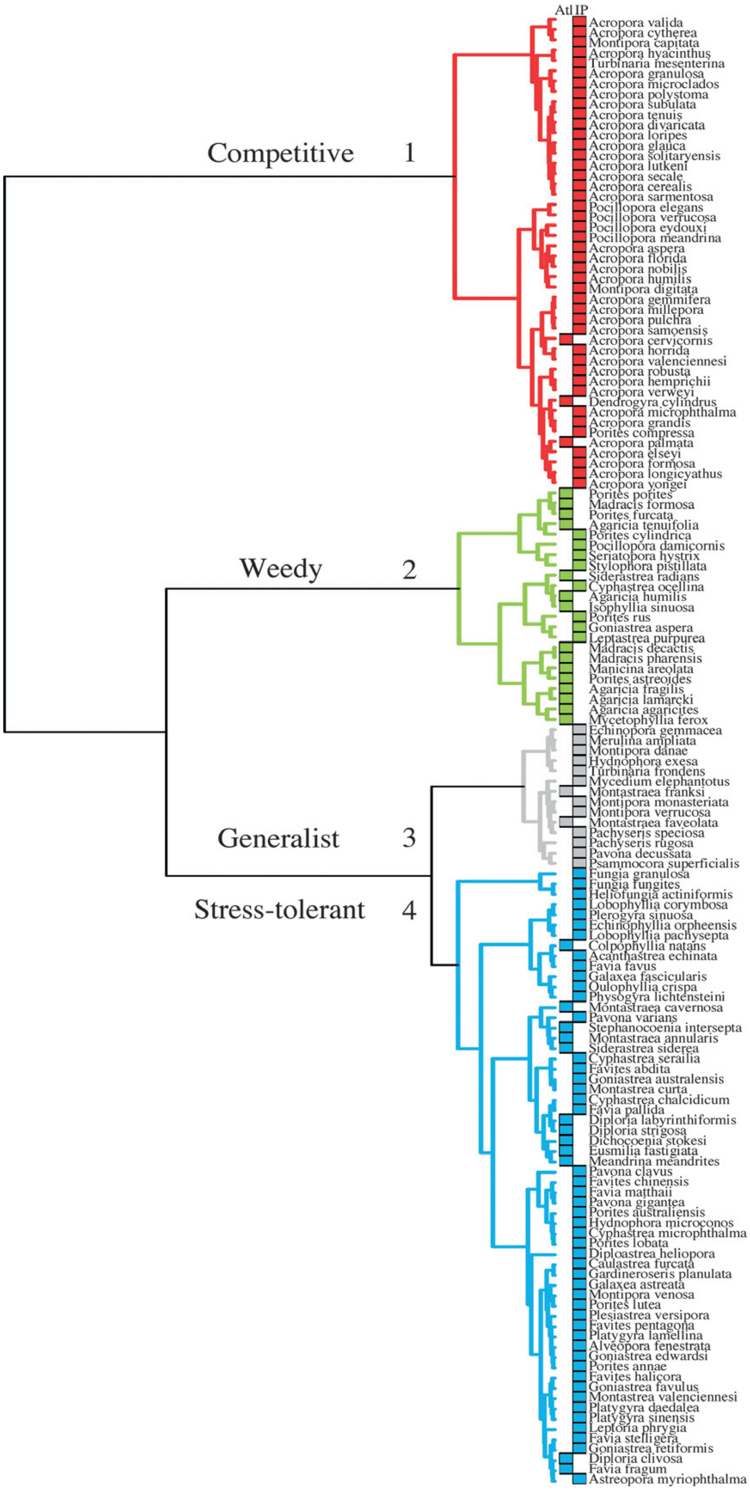


Figure 4.3: Hierarchical cluster analysis of global scleractinian corals based on species traits. The analysis includes 143 species with $\geq 60\%$ trait coverage (i.e., information for at least 7 of 11 traits). Boxes indicate Atlantic (Atl) or Indo-Pacific (IP) distributions. Description and image by Darling et al. (2012).

Relevance to practice

While the descriptions based on traits in Darling et al. (2012) are interesting, these are primarily useful to get a first rough impression of a likely change in community dynamics when adding additional exposures to this community of species. Besides the already mentioned problem with discretizing in a number of groups (as opposed to a continuous gradient), large variations within coral species have been observed (and are not limited to one of the earlier mentioned groups), which could easily lead to classifying a species into a different group. With that said, the authors of (Darling et al., 2012) do hypothesize that increasing disturbances or stress lead to a less abundant presence of the first 'competitive' group; some examples of shifts from 'competitive' dominated to 'weedy' or stress-tolerant dominated have been found in practice (Darling et al., 2012, and references therein). Even though not discussed for mangroves and only briefly for seagrass, similar assumptions can be made for these two groups as well.

4.1.2 Variations in species

Species in general have differences, and individuals belonging to the same species might also show (though relatively smaller) differences. This might seem like a trivial point to make, but it makes it almost impossible to assign a single value to a certain attribute and still be accurate. In the case of differences between species, this could be solved by assigning separate values for each species, but it would almost certainly be too laborious to assign a specific value for each individual. In this case, it might be beneficial to capture the variability of a parameter in a stochastic variable. Both variations between and within species are further explained in the case of corals.

Briefly touched upon in the previous section, significant variations have been found in coral morphology. Between species, growth forms can differ significantly (see also Section C.1). Different growth forms are believed to be associated with specific, different thresholds to exposures. This can already partly be found in Section 4.1 but mention of this was also made in Erftemeijer et al. (2012).

Within species, differences in phenotype can also be found. This variation can have a genetic and/or an environmental cause. A morphological variation of a species whose appearance is determined by its ecological environment is called an ecomorph (Veron, 1995; Larsen, 2012). Genetic variation means that different colonies of the same species have different genotypes. Environmental variation can likewise lead to changes of the genotype, (e.g. due to selective breeding). For example, in relatively low light intensities corals tend to flatten in order to expose as much surface as possible to irradiance (Anthony et al., 2005; Stambler and Dubinsky, 2005; Ow and Todd, 2010; Larsen, 2012). High flow conditions, either due to strong wave action or strong currents, result in slower growth rates and more dense/stronger skeletons (Brown et al., 1985; Scoffin et al., 1992; Larsen, 2012). An example of the difference between high and low flow conditions can be seen in Figure 4.4. A more comprehensive description can be found in Appendix C.

4.1.3 Stable states

The theory of alternative stable states comprises of the notion that for the same set of conditions, multiple stable states are attainable (also known as hysteresis). In the case of a phase shift, the system is perturbed to a threshold which induces a (relatively sudden) shift from one stable state (or phase) to another. An example can be found in the case of coral reefs where macro-algae are abundantly present as well. They compete with corals for, among other things, substrate, nutrients and light (Odum and Odum, 1955). Should coral cover decrease for some reason, it has been observed that a phase shift from coral dominated to algae dominated can occur (Dubinsky and Stambler, 2010; Larsen, 2012). Once in this new state, bringing conditions back to what they where when the reef was coral dominated might not be enough to actually return to a coral dominated state: considerable better conditions could be required to move the system back to the old state.

4.2 Effect characteristics

Attempting to characterize effects shows a number of parallels with life history theory: effects are in fact a continuous gradient, but are often characterized in a limited number of discrete effect classes either for ease of understanding or lacking precision in relevant data. Yet, differences with life history theory can also be found: in life history, traits of species were examined, while characterizing effects explicitly looks at connecting exposures to effects.

4.2.1 Effect classes

Based on personal communication[1] and PIANC (2010), a general classification of effects for corals and seagrass would be:

1. Growth/Positive effects
2. Physiological changes and/or physical variability
3. Disease/temporary damage
4. Mortality

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[1] Experts at DHI Singapore and dr. Johan Stapel from Wageningen IMARES

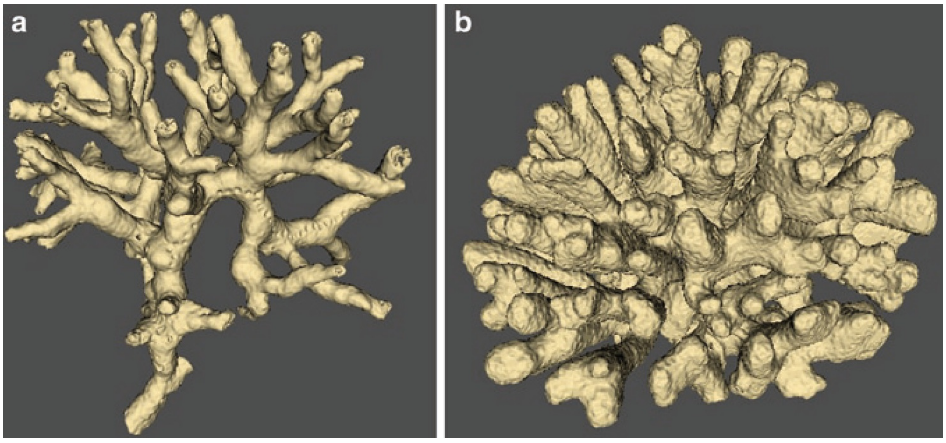

Figure 4.4: Surface renderings of CT scans of a coral species: (a) lateral view of a thin-branching low-flow morph, (b) top view of a compact high-flow morph. Both colonies are visualized on the same scale. Description and image taken from Kaandorp et al. (2011).

Generally, dredge plumes are considered to be the cause of primarily negative effects; this explains the a-priori finer distribution of effect classes for negative effects. However, in the case of mild exposures, beneficial effects might be possible as well. If bleaching due to high light/temperature is expected, shading through elevated TSS levels might be beneficial: it should be noted that this example is currently only found to be an assumption at best and only for a limited set of conditions, see also Appendix E. Attention in many dredging projects is focused on preventing negative effects, but at least a single positive effect class should be included as well. Besides painting a more complete picture when classifying field data, it can also serve as a sanity check between a-priori expected results and obtained model results.

The second response class is considered to be 'always active', which coincides with one of the statements in Chapter 1: there is no equilibrium in the environment, at best it approximates a (temporary) equilibrium. This response class could be seen as a synonym for no response, even though it would be better to describe this class as 'no significant variation compared to naturally occurring variation'. The third and fourth response class are more relevant, because these generally need to be avoided in practice; but also because these occur less frequent during 'normal' conditions and are easier to recognize and measure.

The third response class specifically should also contain any significant reduction in biological parameters; for a list of commonly measured biological parameters see Subsection 4.2.2.

The above is applicable for seagrass and coral; as a very rough rule of thumb, it is often assumed seagrass is able to tolerate twice as much dredging related exposure as coral. Mangroves could be classified in the effect classes physiological, losing leaves/significant less seedling settling and mortality. Mangroves, to a certain degree, need sediment (Subsection 2.5.1), and have high tolerance limits: considering the subject of this thesis and the mangrove tolerance limits which are usually set by (partial) burial of the roots. In practice this comes down to mangroves almost never being the 'most sensitive' receiver in a system where coral and/or seagrass are present as well.

4.2.2 Measuring effects

The list of effect classes in Subsection 4.2.1 indicate primarily biological changes; measuring biological parameters seems therefore fitting to characterize the status and trends in coral reefs and seagrass. According to Hill and Wilkinson (2004), some frequently measured biological parameters include:

- Percentage cover of corals (both live and dead) and sponges, algae and non-living material
- Species or genus composition and size structure of coral communities
- Presence of newly settled corals and juveniles
- Extent and nature of coral bleaching
- Extent and type of coral disease

These are specific parameters for coral, but analogues can be found for the above mentioned parameters for seagrass and mangroves (with the possible exception of coral bleaching). Popular methods for measuring these in the field include usage of divers or underwater equipment to measure one or multiple of the above mentioned biological parameters. Remote sensing is currently used primarily as a tool for site mapping: finding relevant sites. Once located, divers or underwater equipment are used to map the site in greater detail (Hill and Wilkinson, 2004). For measuring exposures such as sea surface temperature, remote sensing can also be used (see Subsection 4.3.1 further on in this chapter).

Measured coral cover percentages are one of the few commonly measured parameters in field data across different projects regarding coral reefs. Coral cover can only be used to detect, given some natural occurring variability, significant (semi)permanent changes in coral cover. This would almost automatically place this kind of data analysis in the first, third and/or fourth response class of Subsection 4.2.1, because the second response class concerns more subtle changes. A popular method for measuring coral cover percentages is the Line Intercept Transect (LIT). This method involves a tape which is stretched closely above the bottom; divers then proceed by recording growth forms (species if possible) directly under the tape. The location can either be random or a permanent measuring placement. LIT can be expanded by using quadrats; a quadrat is a square frame placed on top of a coral reef or seagrass field to sample the area instead of only measuring under the tape (Hill and Wilkinson, 2004).

4.3 Exposure-effect characteristics

As mentioned in Section 4.2, effects can be characterized using measurable biological parameters. The scale at which a problem is perceived determines the requirements for a particular quantification of an effect; and how to model (and thus predict) effects based on expected exposures. Some exposure-effect relations in this section can be called ‘background exposures’; these represent viable quantifications of exposures which are not necessarily influenced by dredging operations, but are recognized to have a significant influence on sensitive receivers. The other exposure-effect relations contain proposed models for modeling known direct dredging-influenced exposure-effect relations. While the number of background exposures and models described in this section is not exhaustive, the exposure-effect models that are in this section were picked either for being commonly used or which appear to be readily applicable in practice.

4.3.1 Temperature induced coral bleaching

Temperature has not been found to be influenced by dredging, but especially corals are known to be sensitive for temperature changes, sometimes even leading to bleaching and mortality (thus can be called a background exposure). Coral bleaching is the expelling of the symbiont zooxanthellae from the coral host due to stress. One of the more prominent stresses associated with bleaching is (aforementioned) temperature. In the case of temperature, either maximum short-term heat stress or cumulative temperature stress is measured using sea surface temperature measurements which are then coupled to previous bleaching events; these results are used to predict future bleaching events. This is model which works on a regional to global scale and provides primarily relative predictions. For a more detailed description regarding bleaching of corals, see Appendix C; for a more detailed description regarding temperature induced bleaching, see Appendix E.

4.3.2 Concentration ratios

One of the most straightforward methods (interpretation wise) available is a ratio between a Predicted Environmental Concentration (PEC) and a Predicted No Effect Concentration (PNEC): if the ratio PEC/PNEC is greater than one, effects are predicted to occur. Disadvantages are quantification of effects and the risk of these effects: if the ratio is larger than one, the only information available is that effects will occur, but not which specific effects will occur and with what probability.

Besides that, PNEC is a conservative number because it is based on end points of laboratory tests, divided by an assessment factor. Uncertainty associated with point estimates of PNEC and PEC can be caught by expressing PNEC and PEC as stochastic variables.[2] PNEC also implicitly carries the notion of a no effect over an indefinite time period; this is a (perhaps too) conservative approach when specifically looking at dredging exposures which usually occur only for a limited time period which are variable (as opposed to constant) as well; The combination concentration-duration is not solved here. Some of these limitations can be found in the fact that these models were originally used in **toxicological** modeling, while dredging exposures are generally not considered toxic.

Exception being if the suspended sediment in suspension carries toxicants, but the suspended sediment itself is not considered as toxic for the sensitive receivers. A more statistically promising characterization would be explicitly stating the percentage affected by a concentration (and have a range of different percentages), but this still does not solve exposure-duration problems or variable concentrations.

4.3.3 Species Sensitivity Distribution

Another problem with PEC/PNEC values is that they are often species specific. If they are used for communities, they first effects will (inevitably) appear in the most sensitive species first. This makes it a rather conservative approach. A logical step would be to somehow show PEC/PNEC values for different species. This is done in Species Sensitivity Distributions (SSD's), and shown in Figure 4.5.[2] While the insight in the distribution of species is admirable, the inherent problems with PEC/PNEC are largely untouched and limitations also apply to this method.

The biggest advantage is the explicit showcasing of effect thresholds for different species, along with the percentage (fraction) of the community each species represents. However, if the data points are based on PNEC values, the disadvantages of PNEC values still hold. Specific to SSD's, the following points are not properly represented:[2]

- Interactions of species are not present
- The notion of 'key species' is not included; if one of these species disappears it could lead to the collapse of an entire ecosystem
- Ecosystem functions are not represented (more on ecosystem function in Chapter 5)

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[2] Building With Nature, "Species Sensitivity Distribution" `http://publicwiki.deltares.nl/display/BWN/Tool+-+Probabilistic+effect+analysis+-+The+Species+Sensitivity+Distribution+-+General` and sources cited within, accessed 01/11/2012

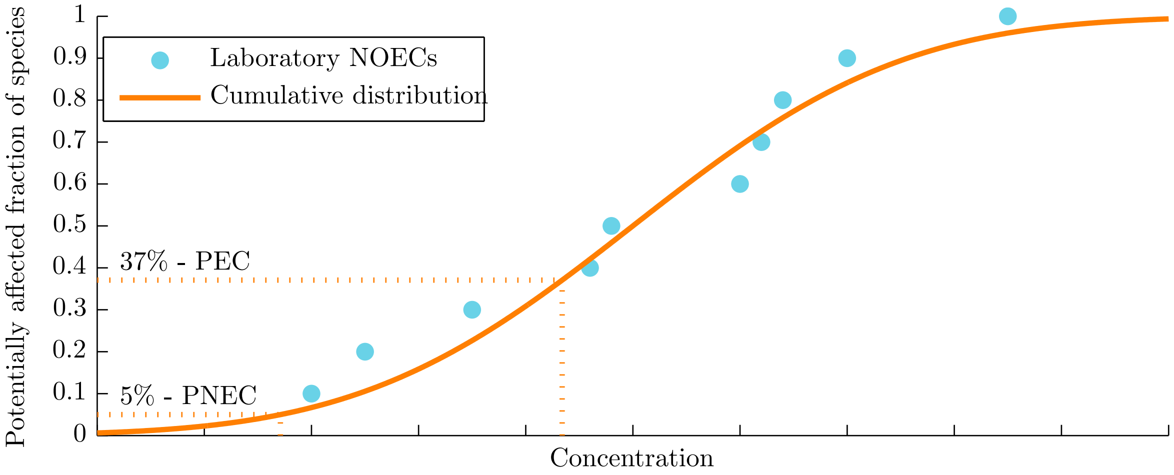


Figure 4.5: Hypothetical SSD for ten species, each representing 10% of the total. In this graph, PEC is chosen as the level that affects 37% or more, while PNEC is chosen as affecting 5% or less of the species. Image adapted from Building With Nature, "Species Sensitivity Distribution" `http://publicwiki.deltares.nl/display/BWN/Tool+-+Probabilistic+effect+analysis+-+The+Species+Sensitivity+Distribution+-+General`, accessed 01/11/2012.

4.3.4 Threshold tables

Threshold tables, in the context of dredging engineering, are tables containing thresholds for spill budgets (often) combined with duration limits for these spill volumes. These tables with spill budgets/durations are linked to effect quantifications; these quantifications are often given in classes of mortality, e.g. 25% mortality, 50% mortality; an example can be found in Table 4.1. An advantage over PEC/PNEC is that there are a now a number of exposure-durations; but the number of options is still discrete/finite. Furthermore, underlying used models are not always obvious and datasets are not always public or rely on expertise only available with the company advertising such tables. Highly variable signals with a mean concentration equal to a constant concentration signal with a similar mean value, also have the same effect according to these tables; which seems a simplification.

4.3.5 Ordered Response model

A commonly used method in toxicology to describe exposure-effect relationships is the probit (or logit) model. (In toxicology these are called dose response relationships, but in the following these are called exposure-effect relationships.) The foundations of this model is the assumption of a relation between exposure effect is shaped like a cumulative normal distribution (or cumulative logistic distribution for logit). in its basic form, given certain exposure(s), only a binary response is given: effect is present with chance X, or it is not present with chance $1 - X$.

Exposure (or dose) is not limited to a single type of exposure; multiple exposures can be combined into a representative exposure. Effect does not have to be binary either: when an ordered response model is used, an arbitrary number of responses can be defined (for instance linking the responses to the effect classes of Subsection 4.2.1). An example of data-fitted ordered probit model can be found in Figure 4.6. A more detailed description can be found

Severity	**Definition (excess concentration)**
No Impact	Excess Suspended Sediment conc.$>$ 5mg/l for less than 5% of the time
Slight Impact	Excess Suspended Sediment conc.$>$ 5mg/l for less than 20% of the time Excess Suspended Sediment conc.$>$ 10mg/l for less than 5% of the time
Minor Impact	Excess Suspended Sediment conc.$>$ 5mg/l for less than 20% of the time Excess Suspended Sediment conc.$>$ 10mg/l for less than 5% of the time
Moderate Impact	Excess Suspended Sediment conc.$>$ 5mg/l for less than 20% of the time Excess Suspended Sediment conc.$>$ 10mg/l for less than 5% of the time

Table 4.1: Example severity matrix for suspended sediments on corals in environments with high background concentrations. Description and table by (Doorn-Groen and Foster, 2007).

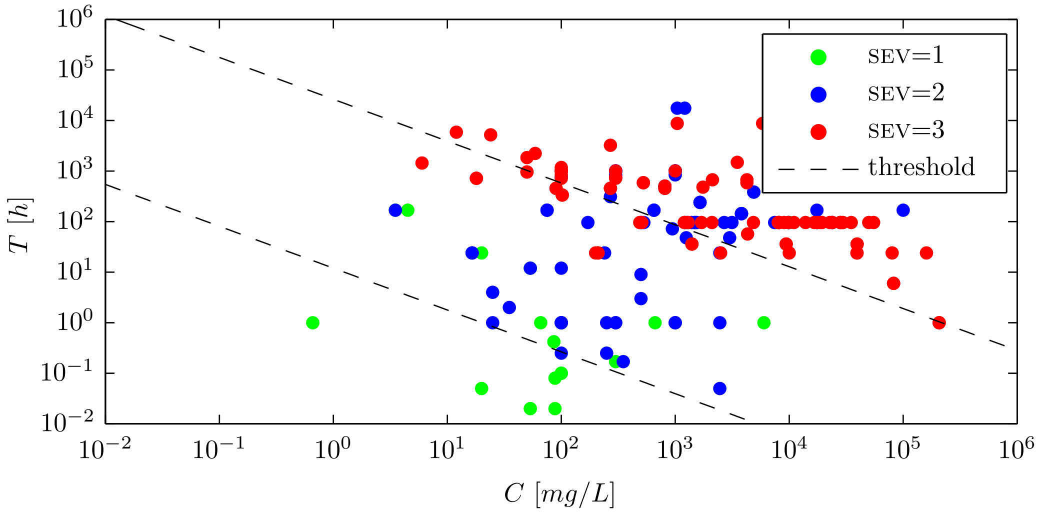
in Appendix A; a complete description can be found in Becker (2011).

There are some remarks to this method as well.

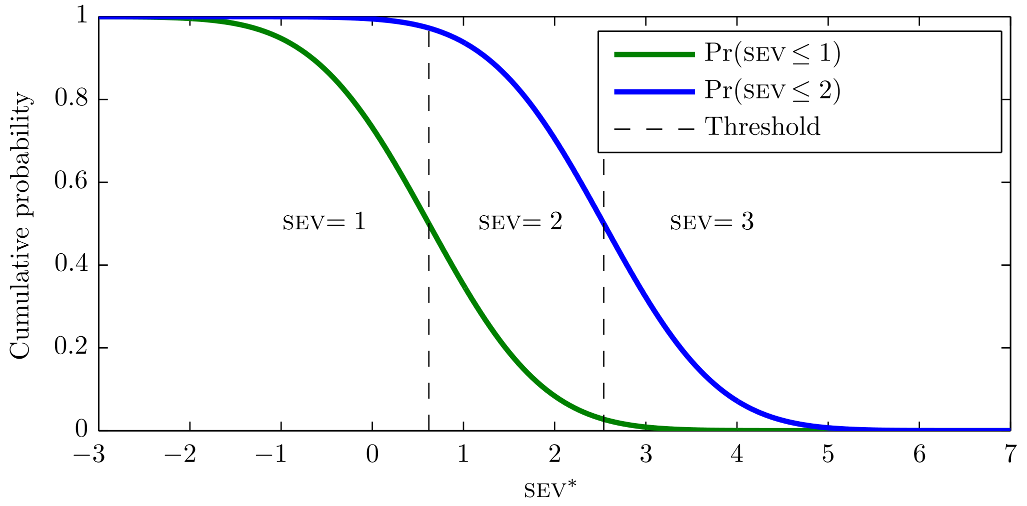
- Describing the exposure-effect relation with a normal (or logistic) cumulative distribution function is an assumption which has not yet been verified for ecological modeling.
- Variable concentration signals can be used; though this includes integration of the concentration signal over time. The concentration signal can be given extra (or less) weight by raising the values in the concentration signal to a certain power. If the power is greater than one, extreme values will become relatively more extreme; if the power is smaller than one, extreme values will become relatively less extreme. For a more detailed description, see Appendix A.
- However, because of the integration, any timeperiod where there is zero concentration is ‘lumped together’; if a concentration signal is either zero or 5 and the total area under the chart is the same over the same timeperiod, one large break with zero concentration would result in the same integrated answer as 10 small breaks with zero concentration. While this may be an implicitly made simplification, *highly* variable exposures might not be represented accurately by this.

4.3.6 General Unified Threshold Model of Survival

The General Unified Threshold Model of Survival (or GUTS) is a framework that describes a number of Toxicokinetic-toxidynamic (TKTD) models; a common feature of these models is the simulation of the time-course of processes leading to toxic effects on organisms. This is done by translating an external concentration of an exposure to an internal concentration over time. A detailed description of the derivation and approximations of the GUTS model can be found in Appendix B. The biggest advantage over the probit models described in



(a) Scatter plot on a log-log scale with severity of ill effect (SEV) as a function of suspended sediment concentration C and exposure duration T. SEV $= 1$ leads to behavioral effects, SEV $= 2$ to sublethal effects and SEV $= 3$ to lethal and paralethal effects. The threshold values are represented by the two black lines. Description and image by Becker (2011).



(b) Exposure-effect curves after fitting the ordered probit model. The cumulative probabilities $\Pr(\text{SEV} \leq j)$ are plotted as a function of effect class SEV*. The probability $\Pr(\text{SEV} \leq 3)$ is 1 for every value of the representative exposure $x'\hat{\beta}$; threshold values are represented by the two black lines. Description and image by Becker (2011).

Figure 4.6: Data application of the ordered response model done in Becker (2011), details can be found in Appendix A.

Subsection 4.3.5 is the notion of an internal concentration; which allows during extended periods of no external concentration to bring the internal concentration down as well. This potentially allows the model to better predict effects when a highly variable exposure is used; especially when the observed effect is mortality. However, some remarks apply to this model as well:

- Even more so than the probit model, this is a toxicological model. The notion of an internal concentration makes sense in the context of toxicology, but it has not been used in ecological modeling for dredging exposures: elevated levels of sediment can be harmful for sensitive receivers, but considering them toxic might be a step too far.

- Only an internal concentration is modeled and this is linked to effects. In the derivation of GUTS in Appendix B a cumulative distribution function is used, which allows monotonic increases (or decreases if the complement is taken). This means that “refilling the stock” (for example by means of reproduction) is not possible in this model.

- The previous point also makes the application less than ideal for sublethal effects: sublethal effects (such as light damage, or even light bleaching in the case of corals) can go away over time; this is not the case in the model. If for example 20% of the individuals exhibit a sublethal effect at a certain point the percentage can only stay the same or get bigger, not less.

- Making sense in a toxicological environment, an initial internal concentration of the exposure is assumed to be zero. This may not be the case for dredging exposures such as suspended sediment concentrations, which are more frequently already present in the environment. This could be solved by allowing for a “spin-up” time. To a certain extent, this could also be said about the probit model and PEC/PNEC; the influence of background concentrations possibly influences results.

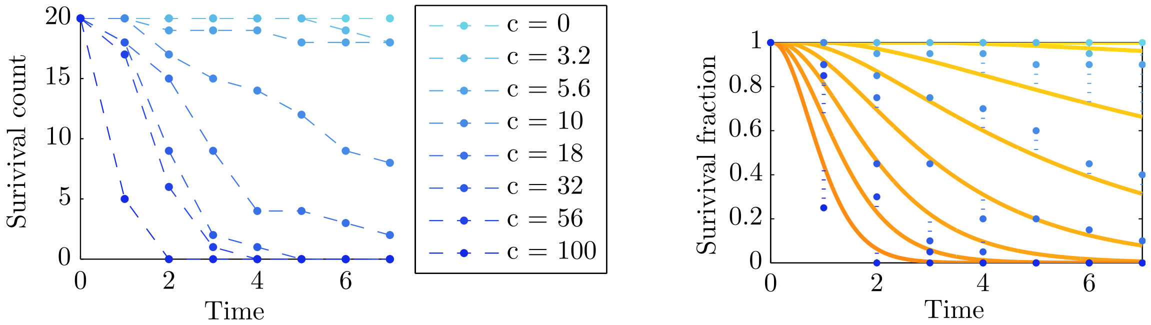
- Contrary to probit, only concentration (allowed to be varying in time) is the only allowed exposure: probit allows for an arbitrary amount of exposures, though probit seems to be tied as well to concentrartion and time when using variable concentrations.

- In the GUTS model, a background “killing rate” is included which is constant over time. For relatively short durations, this can make sense but over longer periods the assumption of a constant killing rate seems to be an oversimplification.

An example of the usage of a GUTS model is shown in Figure 4.7. After calibrating the model on experimental data in Figure 4.7a and Figure 4.7b, the model can be used to predict the response to a variable concentration. The model does this by translating the external concentration to an internal concentration in Figure 4.7c. The fraction which dies for the model shown in Figure 4.7 is then determined by the duration and severity of exceedance of the threshold; this can be seen in Figure 4.7d.

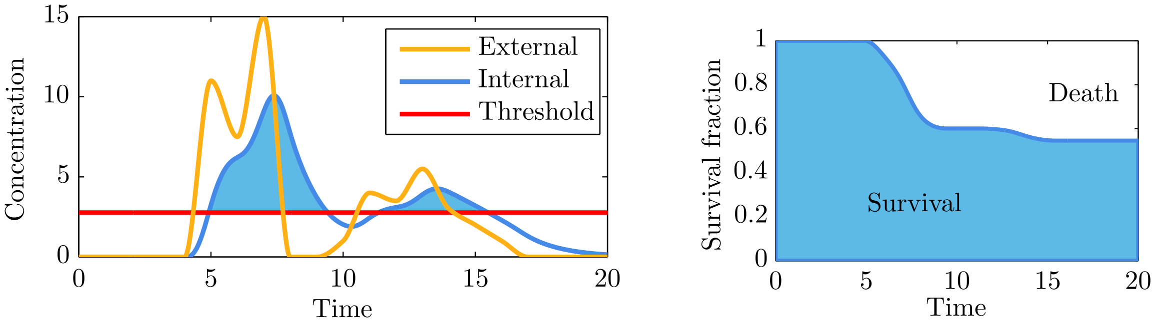
4.3.7 Regional models

It is possible that in a region a single coral reef can provide necessary parts for healthy coral reproduction of other coral reefs in the vicinity. A common way for coral reefs to reach other reefs is by means of spawning: this results in the release of reproductive parts which are



(a) *Data collection:* experimental data of a specie exposed to various concentrations.

(b) *Calibration:* finding a common set of model parameters which have the best fit with the experimental data. If enough data is present, perform *validation.*



(c) *Application:* the calibrated model translates an external concentration to a (scaled) internal concentration. If this internal concentration exceeds the threshold (which is one of the calibrated parameters), death is expected.

(d) *Visualization:* estimate of the surviving fraction due to exposure to the variable concentration.

Figure 4.7: Typical steps in applying a GUTS model, see also Appendix B

transported in a similar manner as the fine sediments in a passive dredge plume: by means of hydrodynamic modeling combined with particle tracking (Melbourne-Thomas et al., 2011b). For a more detailed description regarding reproduction of corals, see Appendix C.

As proposed in Melbourne-Thomas et al. (2011a,b), CORSET (Coral Reef Scenario Evaluation Tool) is a large scale model, put forward as a regional management support tool. It models spawning and brooding corals, macroturf, macroalgae, fish and sea urchins. It models growth, competition and recruitment, and the model is forced with fishing-pressure, destructive fishing activity, damaging storms, coral bleaching disease, crown-of-thorns starfish outbreaks, nutrification and sedimentation. CORSET predicts the relative consequences of different management scenarios on a very large scale (100s of kilometers), and is a very general model (Melbourne-Thomas et al., 2011a,b; Larsen, 2012).

4.4 Energy budgets

Ideally, a formal quantitative description is given of the development of sensitive receivers; this means describing a life of a sensitive receiver and all the significant factors that influence it. An energy budget describes the flow of energy, which can theoretically predict development, respiration and reproduction in all life stages; though this depends very much on the desired detail of the energy budget. Very general broad stroke energy budgets are possible as well; but in the end the basic principle of energy balance underlies it all. Because energy budget models do not focus on a single exposure, growth can be properly accounted for; but one of the (obvious) drawbacks of wanting to describe everything results in needing a significant amount of background information and data regarding the particular sensitive receivers to correctly schematize parameters.

4.4.1 DEB theory

One of these models is Dynamic Energy Budget (DEB) by (Kooijman, 2010). DEB is a systematic but abstract way of describing uptake and usage of energy. It can be influenced by outside factors such as food density and temperature, while predicting/describing the aspects mentioned in the introduction of this section about energy budgets. Furthermore, multiple life stages can be incorporated. However, neither mangroves, seagrass or corals have yet been used in conjunction with DEB in an actual application. This is probably because coral is a difficult organism to model (symbiosis with zooxanthellae, so actually two very different organims need to be modeled including connecting energy flows). For plants, which are primarly autotrophs, energy flows are also harder to model than hetertrophs, because autotrophs can “make” their own food from for example light in the case of light-autotrophs. This makes their food distinctly different than their building blocks (e.g. CO_2, nitrogen, phosphorous).[3]

4.4.2 Coral productivity model

Particular implementations for coral, albeit simplified compared to DEB, can be found in Anthony and Fabricius (2000); Anthony and Connolly (2004); Larsen (2012). Because the model in Larsen (2012) was worked out specifically for (elevated) turbidity and coral, this model is used in the remainder of this section. The model by Anthony and Fabricius (2000); Anthony and Connolly (2004) uses the same fundamental equations, but is worked out differently.

In Larsen (2012), light attenuation is picked as the primary influence for coral health (reasons for this can be found in Appendix C), with the model schematizing this at its core as photosynthetic production by zooxanthellae which is partly allocated to the coral host. A detailed description of corals and the coral productivity model by Larsen (2012) can be found Appendix C. Because photosynthetic production was considered important in Larsen (2012), a description of light and the supposed influence on corals (and with some generalizations seagrass as well) can be found in Appendix D.

The output of the model in Larsen (2012) is primary in terms of carbon content (a proxy for biomass). Growth of corals is schematized by means of an energy budget: input minus the output of energy. Energy input (I) is attained from heterotrophic and autotrophic feeding. Energy output contains metabolism, or respiration (R), mucus excretion (M), carbon sequestration (S, capture of carbon: skeleton growth) and additional outputs such as reproduction and predation (B). This is summarized in a general equation in [4.2], where energy is expressed in terms of organic carbon (C) (Larsen, 2012). A graphical representation is shown in Figure 4.8.

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[3] Personal communication with Tjalling Jager, Department of Theoretical Biology, Vrije Universiteit, de Boelelaan 1085, NL-1081 HV, Amsterdam, The Netherlands

$$\frac{dC}{dt} = I \cdot C - R \cdot C - M \cdot C - S \cdot C - B \cdot C \qquad [4.2]$$

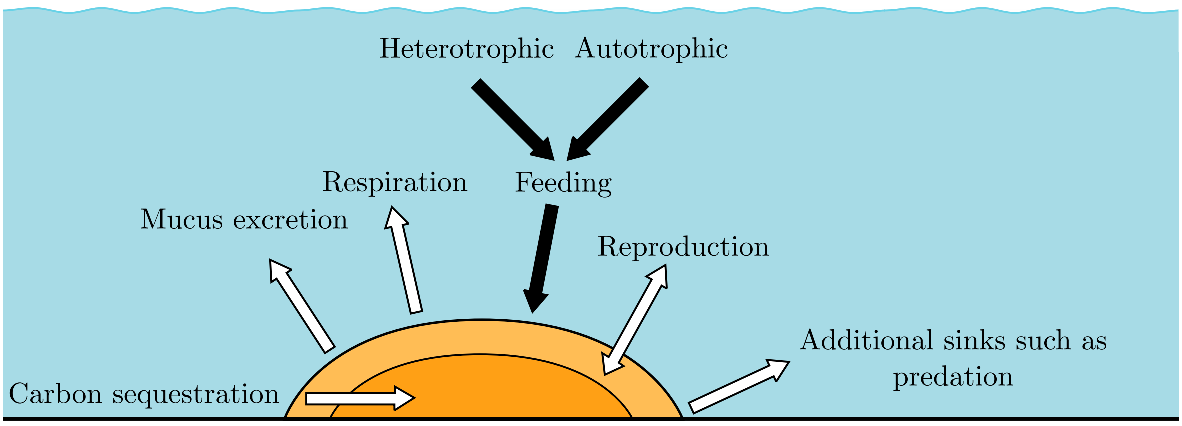

Figure 4.8: Graphical representation of equation [4.2]. The light arrows indicate energy output, the dark arrows indicate energy input. The light arrows that are pointed inwards indicate an eventual contribution to growth of the coral.

Some general remarks regarding these models:

- Usually a large number of processes are described in energy budget models which are understood in theory, but often lack accurate estimates of practical values. This can be a problem especially with a large number of variables.
- Receiver size or biomass is chosen as the parameter to be modeled in energy budgets. This makes it suitable for modeling effects that can be related to changes in biomass.
- Especially where growth is expected to play a significant role (i.e. low exposures), energy budget models are probably better suited to solve exposure-effect relations than models originating from toxicology which, compared to dredging, often deal with shorter periods, higher exposures which show less variability.
- Because of the relative large number of variables involved, a Monte Carlo simulation was done for the coral productivity model by Larsen (2012), which can also be found Appendix C.

Chapter 5

Risk assessment

Before a formal probabilistic calculation can be made as mentioned in Section 2.2, the dredging operation has to be planned and sensitive receivers have to be identified; a global overview of how this could be approached is given in Section 5.1. During the execution phase, described in Section 5.2, adjustments can still be made although these are more restricted than during the planning phase. Further on in this chapter, a qualitative description is given for making the choice for exposure-effect quantifications (Section 5.3). Finally, some consideration is given to valuation of a dredging project in Section 5.4.

5.1 Planning and identification

5.1.1 Identifying sensitive receivers

Given a project location for a dredging operation, a system boundary has to be drawn; a starting point for a suitable boundary would be the area where a possible passive sediment plume has no significantly different concentrations compared to the presumed background concentration. Within this area, sensitive receivers are then identified. Depending on the project (budget, time, requirements) investigations could be made regarding regional influence of for example coral reefs (as described in Subsection 4.3.7), which could possibly expand the previously identified system boundary because of regional interdependence.

At this point, mangroves have been only mentioned occasionally in this document: this is because in general, mangroves are often found in systems where coral and/or seagrass are present as well. If coral or seagrass are present, general belief is that coral and seagrass are ‘more sensitive’ than mangroves for dredging exposures such as increased sedimentation and suspended sediment. Qualitatively, it can be said that mangroves, being adapted to a dynamic sediment environment are able to thrive in different sedimentation regimes and can thus cope with sediments better than other receivers.[1] Quantitatively, some preliminary numbers for sedimentation can be found for mangroves in Subsection 2.5.1 and for coral in Section C.3.2, which indicates that coral indeed is more sensitive. In other words: coral and seagrass reach their limit before mangroves get uncomfortable. This does not mean mangroves should be neglected: in the case where only mangroves are affected or for cases outside the scope of this document (polluted sediment, very close to the dredging operation) mangroves especially

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[1] As discussed at DHI Singapore

need to be evaluated. In the case of large expected sedimentation, especially fine sediments, concerns regarding root stability for both seagrass and mangroves should be contemplated.

5.1.2 Planning

Once sensitive receivers are identified, relevant baselines need to be established. These could warrant measurement surveys and/or re-using existing datasets (if accurate and recent enough). Some relevant metrics were mentioned in Subsection 4.2.2 for measuring receiver health:

- Percentage cover of corals (both live and dead) and sponges, algae and non-living material
- Species or genus composition and size structure of coral communities
- Presence of newly settled corals and juveniles
- Extent and nature of coral bleaching
- Extent and type of coral disease

And some frequently measured parameters related to the physical environment are (Hill and Wilkinson, 2004):

- Depth/bathymetry
- Currents
- Temperature
- Water quality
- Visibility
- Salinity

Some of the parameters are measured by hand or are assessed by divers, making them inherently prone to inaccuracies and uncertainty: see also Subsection 2.3.1. When risks are discovered that are not acceptable, changes can be made to the planning. There are several options to reduce dredging exposures which can be categorized into four groups (Bray, 2008; Becker, 2011):

1. Use of different equipment
    (a) Environmental disc bottom cutter
    (b) Sweep dredger or low turbidity dredger, which are useful for environmentally sensitive projects
    (c) Auger dredger, which is especially suitable for clean up
    (d) Environmental grab
    (e) Anti Turbidity Valve (ATV)

2. Change in way of operation
   (a) Speed of the vessel
   (b) Pump speed
   (c) Navigation
   (d) Overflow discharge
   (e) Hoisting speed
3. Environmental windows which restrict operations only during certain time slots. This could be related to (for example) seasons and/or tides.
4. Physical barriers such as a silt or bubble screen

Once a first assessment has been made of the kind and species of sensitive receivers, some approaches to get a better grip on what globally the effect of exposures is going to be on the investigated sensitive receivers. For example, life history Subsection 4.1.1 could give a very crude first estimate on the effect (and possible relative recovery speed).

5.2 Execution

During execution, there is not enough time for predictions and measurement campaigns such as mentioned in Section 5.1. If during the planning phase a probabilistic calculation was made, the results of this calculation can be used in a deterministic calculation if conservative values of the probabilistic calculation are used. This can be done by picking a (for example) 95% confidence border from the parameter's distribution (Becker, 2011); see also Section 2.2. Particular for exposures, as already mentioned in Section 3.5, instead of a full probabilistic calculation with every possible situation, only certain likely scenarios could be modeled.

In (PIANC, 2010), it is recommended that an Environmental Management Plan (EMP) is continuously updated; during execution measurements can be made of sediment spill which can be used to refine or hindcast passive dredge plumes (and compare these with the measurements). Based on these results, and ecological indicators, production of the dredger could be altered/optimized. During execution, several practical things can be done which can serve as some kind of early warning system. For example:

1. In the coral productivity model and in the case of dredging, light is seen as the primary influence for coral biomass (this follows current literature such as Erftemeijer et al. (2012)). The deepest part of a coral reef get the least amount of light: for individuals of the same species, this would imply the lower lying individuals would exhibit effects first. More details regarding this can be found in Appendix D.
2. The notion of "coral sentinels". By transplanting small pieces of corals closes to the dredging source, the effect of higher exposures can experimentally be found. Although this seems to be rather straightforward, care has to be taken that the coral pieces actually can survive in the new location and that conditions are approximately the same. Otherwise, the usefulness of this technique dwindles. Besides practical issues, legal issues might exist as well; e.g. red list species would normally prevent this method.

5.3 Exposure-effect quantification

In Chapter 4 a number of theories and models were mentioned regarding how to model and quantify effects on sensitive receivers due to exposures, each with their own strength and weaknesses. In the past, methods such as concentration ratios (Subsection 4.3.2) were used, but were quickly found to be either too conservative in the estimates given. In search of a better prediction, current dredging practice tries to give estimates of effects using tables which are based on so-called spill budgets or, a more refined, spill budgets per unit of time. These methods however, albeit simple to use, do not give a direct relation between exposure and effect; rather they are endpoints of (confidential) modeling. Two models which were recently introduced attempting to solve this, are the ordered response model (Becker (2011), Subsection 4.3.5) and the coral productivity model (Larsen (2012), Section 4.4).

5.3.1 Relations between exposure and effect

Becker (2011) proposed a method which is based on probit or logit dose-response models, able to describe multiple (ordered) responses as a measure of effect. The basis of this method is the assumption of cumulative normal (probit) or logistic (logit) distribution as the exposure-effect relation; dose and response are seen here as synonyms for respectively exposure and effect. Though this method has a direct link between exposure and effect, the choice for normal or logistic distributions is an assumption (an often made assumption nonetheless); details regarding the implementation can be found in Subsection 4.3.5/Appendix A.

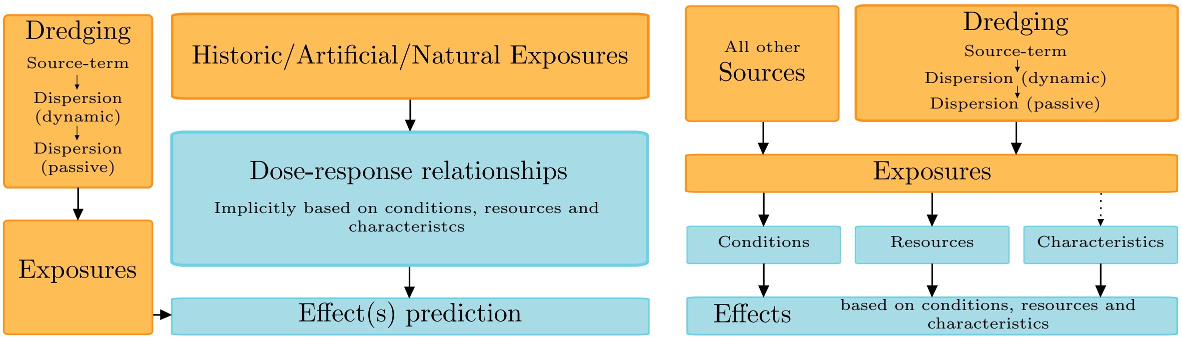
In Larsen (2012), a more process based (bottom-up) approach is taken: processes considered important for coral growth (both negative and positive) are combined in an energy budget approach. The difference between the aforementioned two methods is shown in Figure 5.1.

However, combining processes in an energy budget requires deeper biological knowledge, and parameters of the processes also need proper, accurate values. Often, the number of parameters in energy budgets is much larger than those in the probit ordered response model or GUTS model. Even though the coral productivity model as proposed by Larsen (2012) and detailed in Subsection 4.4.2/Appendix C is rather simplified compared to the detailed, more holistic approach to energy budgets in DEB theory (Subsection 4.4.1), the latter is significantly harder to implement for the current set of sensitive receivers. In fact, no practical implementation for corals, seagrass or mangroves using DEB could be found in literature. The coral productivity model on the other hand has already seen an (preliminary) application in Larsen (2012).

The ordered response model by Becker (2011) uses less parameters, but these parameters need to be calibrated first in either laboratory tests or field measurements under similar conditions as the to be expected dredging exposures. Furthermore, these parameters do not have a clear, direct biological meaning; in the case of the ordered probit model, the calibrated parameters are shape and place parameters for the cumulative normal distributions (see also Appendix A).

5.3.2 Dredging exposure characteristics

The previously two mentioned models are proposed in their respective documents to model dredging exposure-effect relations (particularly suspended sediment concentrations), but here



(a) Currently proposed implementation by Becker (2011). The dose-response relationships represent an assumed statistical relationship between exposure and effect.

(b) Energy budget approach flow-scheme. Exposures result in changes in conditions and resources (and possibly characteristics) which leads to effects. Similar to Figure 2.6.

Figure 5.1: Flow schemes representing flow-schemes of a currently proposed model (Figure 5.1a) and a schematic of reality (Figure 5.1b). The dotted connection between 'Exposures' and 'Characteristics' in Figure 5.1b is there because it is assumed that exposures only result in structural changes in characteristics on considerably longer timescales (evolution changes characteristics). Given the timescales of dredging projects (commonly ranging from months to a couple of years) characteristics can be considered as constant for a particular place and time.

a case is made that these models are expected to be better in conjunction with certain exposure types. A third model (GUTS, Subsection 4.3.6)is introduced which is believed to be fitting for another type of exposure signal. Backgrounds for these models can be found in Appendix A, Appendix B and Appendix C.

The ordered response model as proposed by Becker (2011), was originally conceived to be calibrated to temporal constant exposures; the effects could then be assigned ordered effects. Using an adaptation to the model, if only concentration and time are used as exposures, temporal variable concentration exposures could also be used (as found in Becker (2011) and sources cited within). This adaptation can be found in Appendix A, and the basic equation is repeated in [5.1].

$$\int [C(t)]^n \, \mathrm{d}t, \qquad [5.1]$$

An integral calculates the surface under a curve; this means as long as the surface under a curve is the same, the predicted effect will be the same. This is part of the assumption, but it can deliver results which are counter-intuitive: integrating three blocks with equal concentration and equal duration would give the same answer as a single block with three times the duration and equal concentration (over the same total time period). Intuitively, these two examples are not the same.

For explicitly dealing with such highly variable exposures, the GUTS framework (Jager et al., 2011) might be a better option as it simulates an internal concentration. Even though it is technically possible to make the GUTS model predict an ordered response (allowing the use of the effect classes as specified in Subsection 4.2.1), the use of sublethal effects only handled well if it is reasonable to assume any sublethal effects appearing during the investigated period do not go away or be ‘repaired’. For lethality, this is a reasonable assumption because resurrecting the dead is not possible.

The effect class growth can technically be included in the GUTS or ordered probit model, but it would require some adaptations because inherent in both methods is the assumption of a negative relation between exposure and effect: greater exposure leads to a greater chance of the effect occurring. The previous leads to believe that the toxicological models of probit and GUTS are best used where the exposure are intense enough to be considered somewhat toxic. If the exposure signal is highly variable, GUTS might perform better judging from the relations used in GUTS.

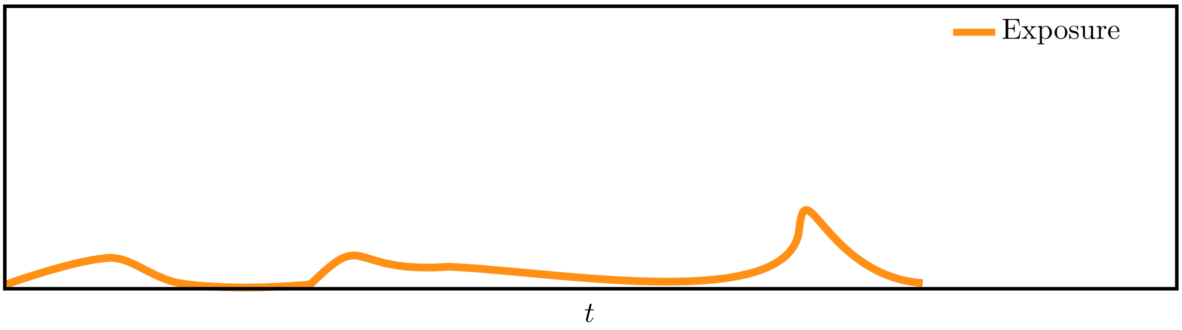
The coral productivity model in Larsen (2012) describes growth (or lack of growth, which in severe cases can lead to decline) of coral biomass. This makes it excellent for scenarios where only light exposures are expected. Also, variability in the exposure signal is not a problem. Intense exposures should not be a problem either, but the large number of variables (compared to GUTS or ordered response) could introduce such an amount of uncertainty to a point where the simpler models of GUTS and ordered response possibly produce better results. An illustration regarding these three exposure types can be found in Figure 5.2.

5.3.3 Receiver characteristics

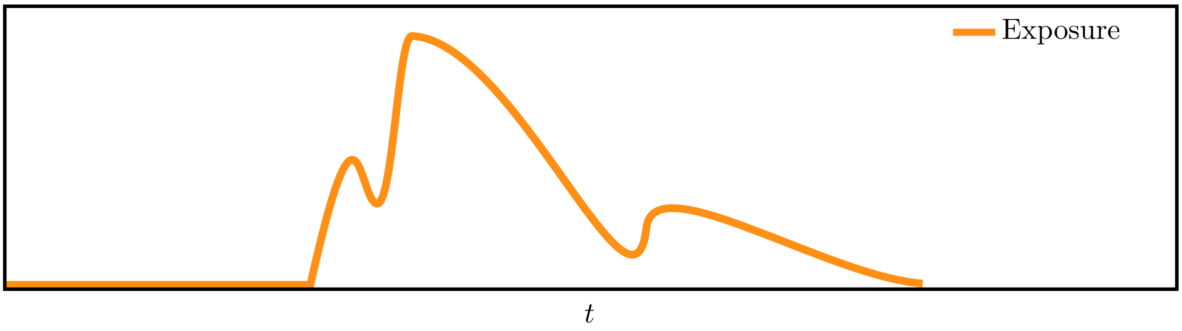
At the basis of all three models is the schematization of an individual. If the example of a coral reef is considered, numerous individuals belonging to multiple species can usually be found. Going from an individual to a species can be modeled in a straightforward fashion: individuals belonging to the same species can be considered to have some inherent uncertainty, but otherwise show similar values for parameters (under similar locations/conditions!). As multiple individuals need to be modeled, spatial differences (or uncertainties) are expected: one of these parameters would be depth which is reviewed in relation to the productivity model in Subsection D.2.2.

A recommendation would be to model different species separately, and portray them using the general idea of a Species Sensitivity Distribution (SSD; Subsection 4.3.3). The usage of PEC/PNEC as mentioned in Subsection 4.3.3 is not recommended; these could be replaced by endpoints of model efforts using any of the above mentioned models.

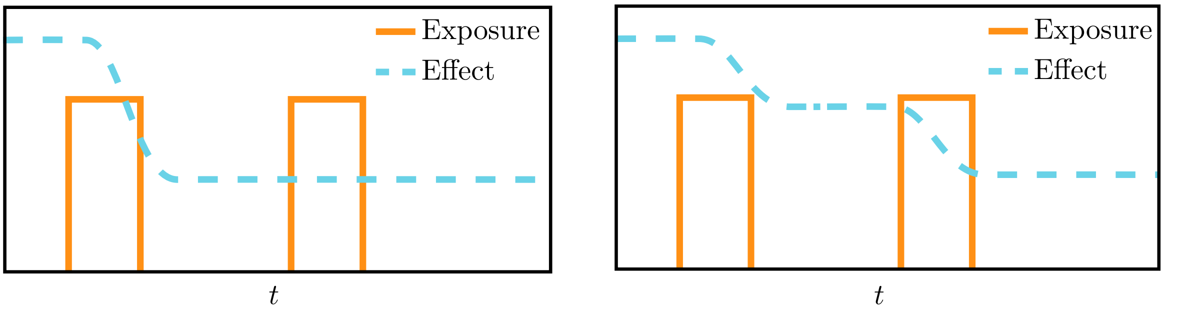
While an SSD might give an overview of the effect and relative size of species in a community, interspecies dynamics are not modeled: certain species might be considered key-species which result in the decline of an entire community if a key-species is removed. Also, phase shifts (Subsection 4.1.3) can occur, either induced due to weakening of the modeled species or changing conditions which increase fitness of a competing species. An example can be found for corals which usually can survive well in low nutrient environments. However if nutrients increase, this could give an advantage to algae. Combined with weakened/dying coral, this could result in a state shift (Richmond (1993); Larsen (2012) and Appendix C). Bottom line, an SSD should not be seen as a representation of a community, rather for what it really is: an overview of the possible effects for a number of separate species. A more global model which is focused on community dynamics can be found in Subsection 4.3.7.



(a) Highly variable, low intensity exposure which is here considered to best fit the coral productivity model by Larsen (2012).



(b) Variable, high intensity exposure which is here considered to best fit the ordered response model by Becker (2011).



(c) GUTS: In this particular form, a stochastic distribution regarding resistance to effects is assumed: the first exposure has effect on the fraction that has not enough resistance. Subsequent similar exposures do not provoke any response.

(d) GUTS: In this particular form, effect is related to the exceedance of the species' resistance. This means that subsequent similar exposures have similar effects like the first exposure.

Figure 5.2: Typical exposure signals which are assumed to best fit a certain type of model. Figure 5.2c and Figure 5.2d belong to the GUTS model and are examples of two different response types for an effect on a (fictitious) population of organisms, both responding to a highly variable, high intensity with relatively large gaps with little to no exposure.

5.4 Valuation of effects

From an economic point of view, a cost-benefit analysis determines if a project is viable: benefits should outweigh costs. While the added economic value of a dredging project is often known, the value of a sensitive receiver (which often is considered to be potentially damaged by a dredging operation) is not clearly defined (yet). Economic value of environmental assets needs to be determined, and can be divided in the following (Ahmed et al., 2005; Becker, 2011):

1. Direct uses (products, recreation)
2. Indirect uses (biological support, e.g. a food source; physical protection, e.g. a salt marsh)
3. Non-use values (option and existence values)

After Barbier et al. (2011), the following list shows an example of services an ecosystem provides, in this case a near shore coral reef. The actual attribution of value is a complex and controversial exercise (Becker, 2011; TEEB, 2010). Without it, it is hard to determine what effects (and to what extent) are actually allowable.

- Raw materials
- Coastal protection
- Maintenance of fisheries
- Nutrient cycling
- Tourism, recreation, education, and research

An example of an attempt to attribute value to (loss of) ecosystem services, in the case of a coral reef, was made in Doshi et al. (2012). Primarily focused on loss of consumer surplus (scuba divers), a bleaching event in 2010 for the sea off Thailand, Indonesia and Malaysia was considered to have cost between 50 and 80 million U.S. dollar (Doshi et al., 2012). Another starting point in attributing value to coral, would be the cost involved in replacing damaged coral with newly bred coral. It is stressed that this replacement cost is not a direct replacement for ecosystem valuation, but it can provide a starting point for valuation. According to Schutter (2010), market size colonies of *Seriatopora caliendrum* and *Seriatopora caliendrum* cost respectively 250-300 and 100-120 euro per kilogram to breed.

Reference situation

Taken from a more philosophical point of view in Chapter 1, and a more practical approach in consecutive chapters, it was implied that by definition there is no stable, tranquil environment where dredging is the only influence. This might seem like a trivial point to make, but it is not trivial to actually apply: often the question asked regarding dredging operations is what the effect of dredging is in an environment. Often emphasis is placed then on the dredging exposure, while it should be a comparison between which (and to what extent) exposures where expected without dredging and which (and to what extent) exposures are expected with dredging.

This implies modeling the reference situation (or: background exposures). In the specific case of coral, this would almost always include monitoring and predicting sea water temperature (Subsection 4.3.1). An article by Death et al. (2012) states several main causes as the decline of the Great Barrier Reef: tropical cyclones, coral predation by crown-of-thorns starfish (COTS), and coral bleaching. In an article concerning mangroves in Florida by Smith et al. (2009), hurricanes and sea-level rise which were hypothesized to have incurred damage and even significant state changes (from mangrove forest to mud flat).

As mentioned in Appendix E, coral bleaching is an increasingly recurring phenomenon for corals, which in extreme cases can lead to extensive mortality, which is primarily attributed to high seawater temperatures but also due to high irradiation. There are combinations of conditions possible where dredging exposure or storms might actually (partially) mitigate bleaching. However, the potentially significant adverse effects of dredging and storms should not be neglected for this reason and the small number of conditions where the influence thought to be beneficial are mostly hypothetical at the moment of writing (see also Appendix E).

Dredging exposures should be seen in context of these exposures, of which the summation is the only reasonable representative in order to quantify effects; this hopefully prevents situations where no significant effects due dredging are predicted, while effects due to other circumstances are blamed on the dredging operation; or where dredging effects are incorrectly put aside as background noise. Besides accurately modeling effects, attributing effects to the correct source is also of major importance in obtaining an accurate valuation of dredging effects.

Chapter 6

Application

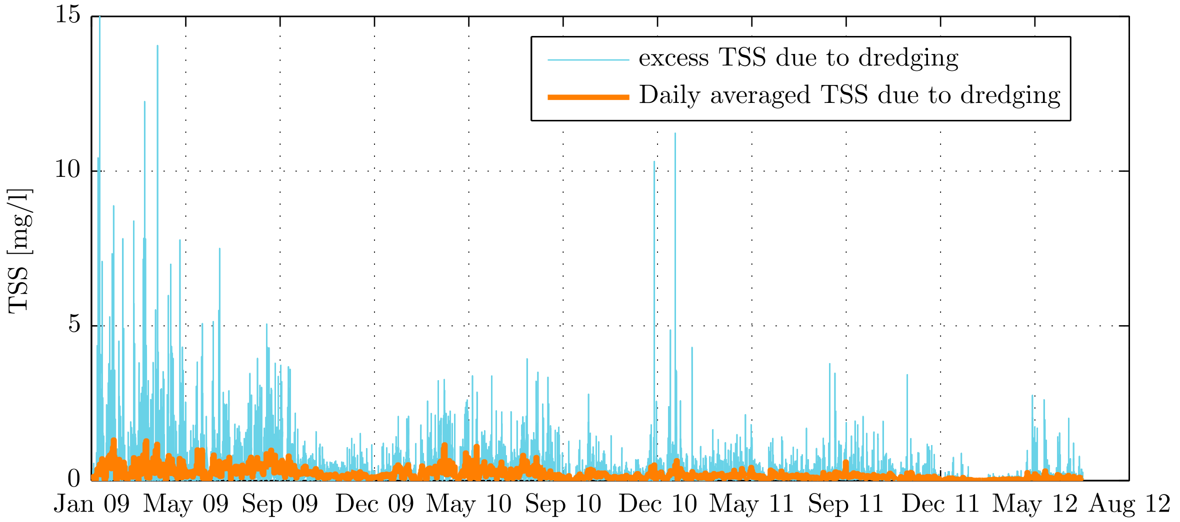
After describing what can be done prior and during a dredging operation regarding ecological modeling of dredge plumes, the following describes a (limited) real world data application. The data is described in Section 6.1, after which an analysis of the data follows in Section 6.2. This chapter is concluded with possible applications of (some) of the models in Section 6.3, as described in Chapter 5.

6.1 Data description

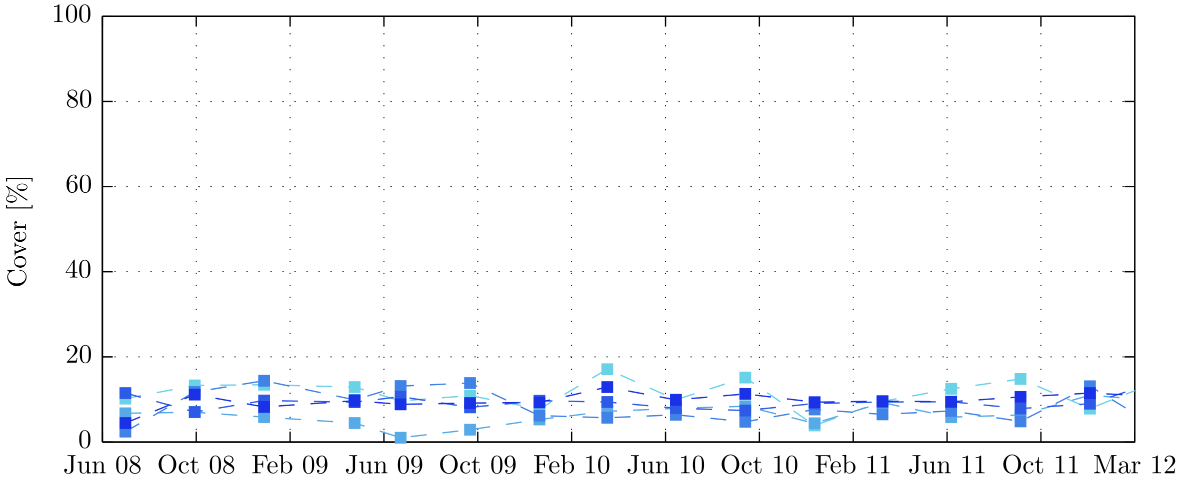
The data consists of hindcasted TSS (Total Suspended Solids) concentrations and coral cover measurements from two coral reef locations in South East Asia. The data was provided by DHI Singapore. The coral cover measurements were done about once every three months using a LIT measurement (Subsection 4.2.2); each measuring campaign was repeated five times. The hindcasted TSS concentrations were made with MIKE 21/3 modeling software; initially predictions, the model was refined using field measurements which resulted in the hindcasted predictions. Figures for both locations (codenames 'CR1' and 'CR2') can be found in Figure 6.1 and Figure 6.2. For ease of comparison, the corresponding subfigures have similar scales.

While the hindcasted TSS can be considered to be quite accurate (apart from model uncertainty), the LIT measurements show large differences even between measurements made at the same locations at the same time. This can be attributed to either the fact that species are not separated, the LIT method and human interpretation (Subsection 2.3.1) or the used metric (coral cover is at best a two-dimensional approximation of a three-dimensional coral structure). At first sight, data does not seem to be strongly correlated, but that is researched further in Section 6.2. Coral cover data is primarily used for management reasons instead of research reasons within DHI; because it serves merely as a check for predicted effects, lower quality data suffices. This made LIT a logical choice (Hill and Wilkinson, 2004).

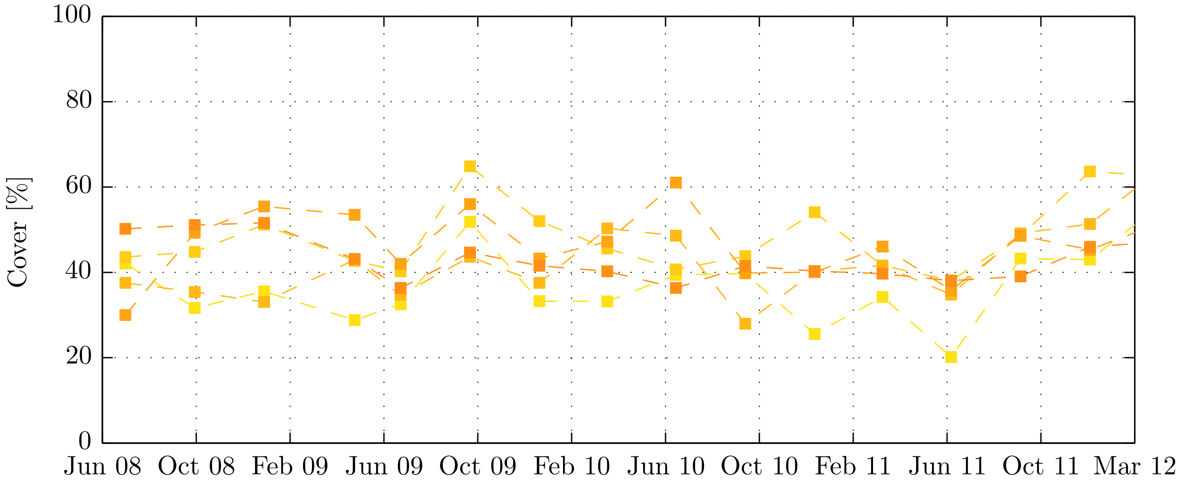
Looking in more detail to the TSS signal in Figure 6.3, it shows rather low daily averages, with temporally short but high concentration peaks instead of a more smoother signal; this can be attributed to the production scheme of the dredging operation or tidal influences. Background exposures were excluded from this research.



(a) TSS levels

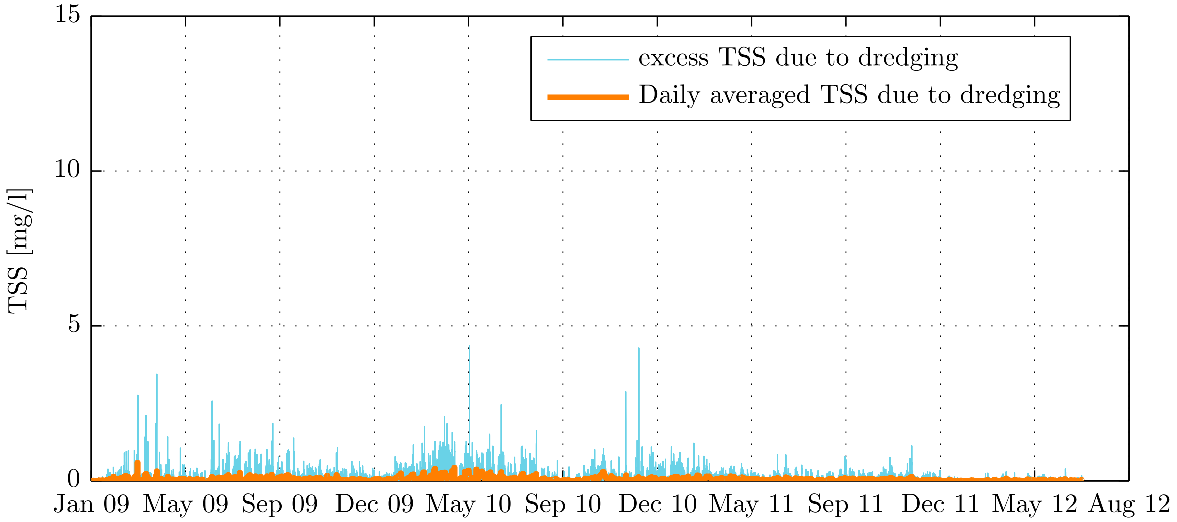


(b) Live coral cover

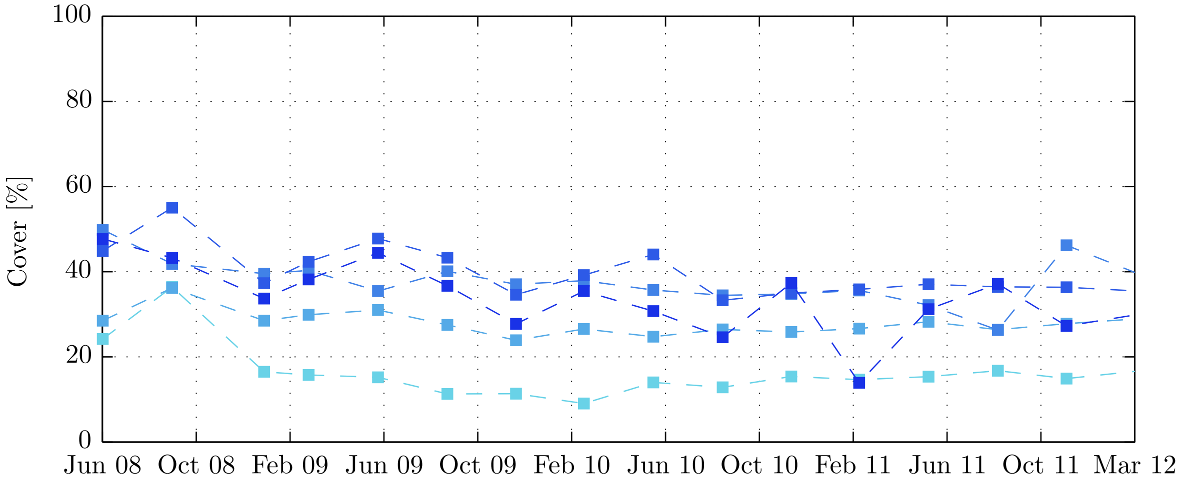


(c) Dead coral cover

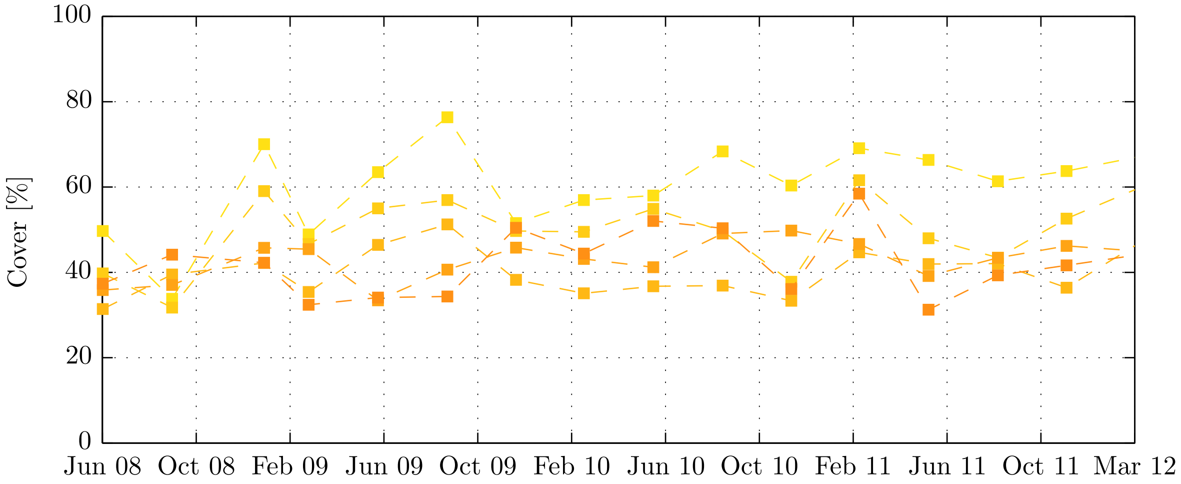
Figure 6.1: TSS and coral cover for location CR1, courtesy of DHI Singapore.



(a) TSS levels

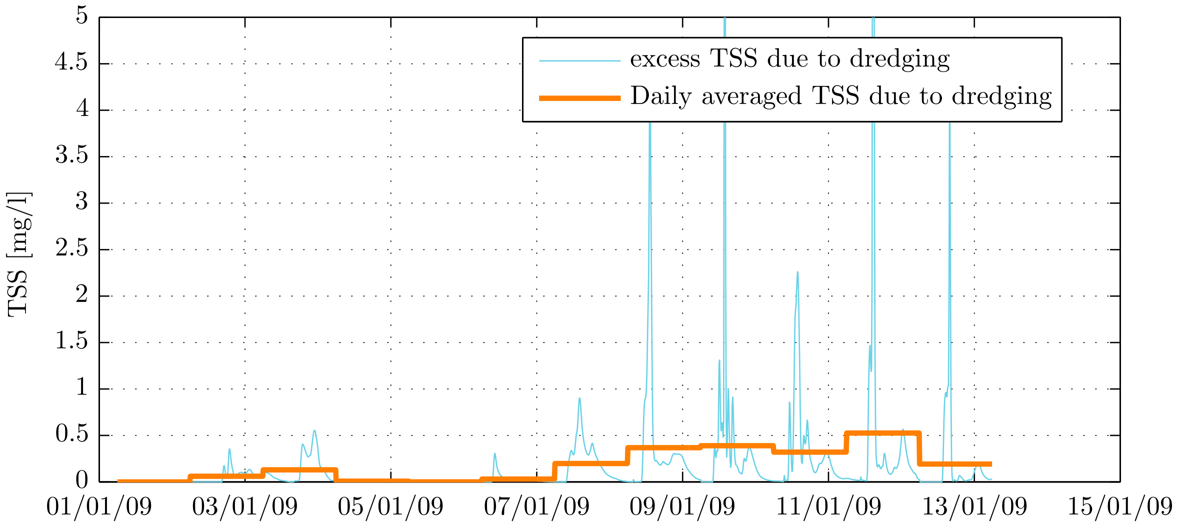


(b) Live coral cover

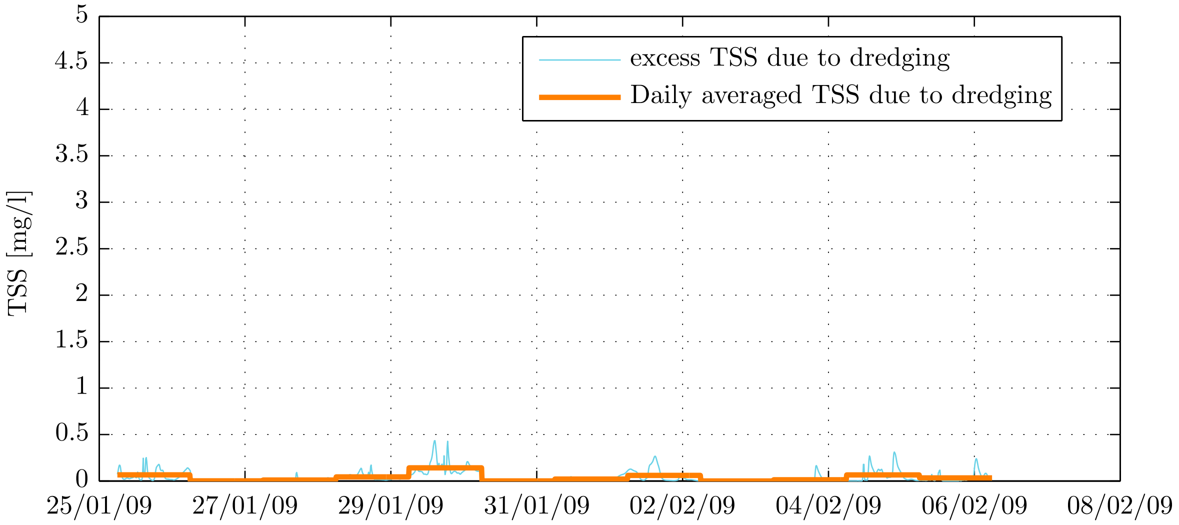


(c) Dead coral cover

Figure 6.2: TSS and coral cover for location CR2, courtesy of DHI Singapore.



(a) TSS levels for CR1



(b) TSS levels for CR2

Figure 6.3: TSS levels for roughly fifteen days in CR1 and CR2. Data courtesy of DHI Singapore.

6.2 Raw data analysis

Because of the low expected correlation on first inspection, a raw data analysis was carried out investigating correlation. For this analysis, some basic transformations were made inspired by the coral productivity model and the ordered response model. In an attempt to couple exposure and effect, the dosefactor equation of the ordered response model was used:

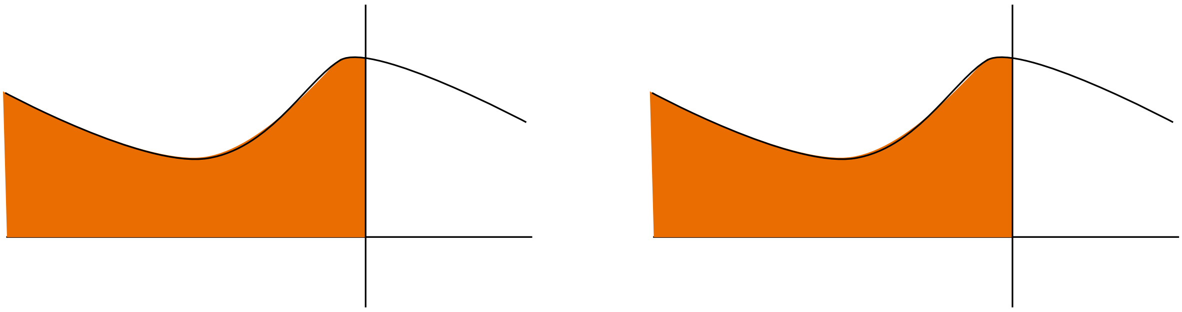
$$\int [C(t)]^n \, \mathrm{d}t,$$

Where n is taken to be 0.5, 1 or 2 resulting in dampening or aggravating extreme values for respectively 0.5 and 2. Different time periods before an effect measurement were taken (the period over which the dosefactor integration goes): either the entire period, a year, 6 months, 3 months, 1 month, 2 weeks or 1 week were considered. Schematizations of some of these combinations can be found in Figure 6.4.

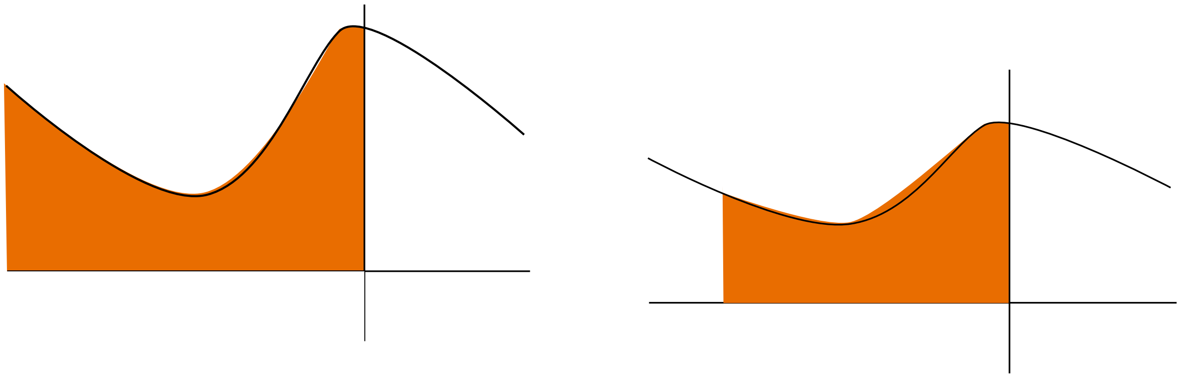
All effect measurements (live coral cover values) for both locations were used. Results of the data analysis can be found in Appendix F (an example of one of these tables can be seen in Table 6.1), and consist of the following seven applications:

1. No time lag, Table F.1
2. No time lag, concentration signal C transformed to $exp(0.01C)$ Table F.2 (inspired by exponential decay of light in water, see Appendix D)
3. Three day time lag, Table F.3
4. One week time lag, Table F.4
5. Two week time lag, Table F.5
6. Four week time lag, Table F.6
7. Eight week time lag, Table F.7

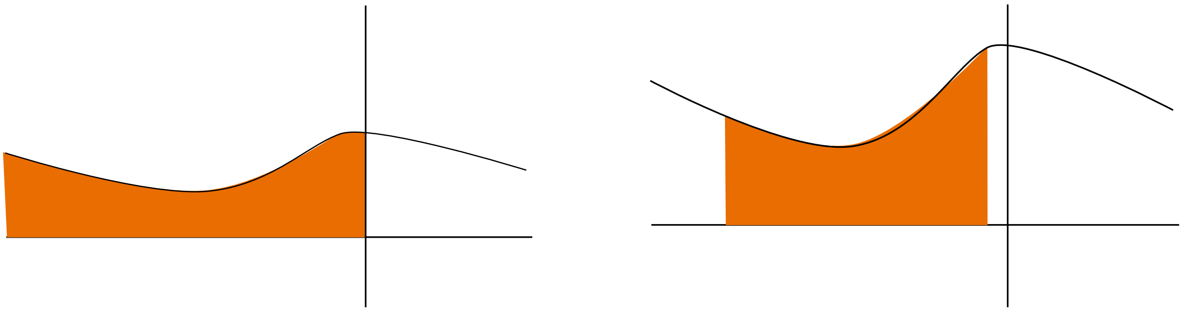
None of the data applications shows a high correlation R. Numbers would be even worse if R^2 values would be calculated. This is actually somewhat expected, because this is data belonging to an actual project; this usually means no significant changes are allowed to occur. Within these results, the exponential transformation did not bring any improvement in correlation. Time lags did not seem to be very influential, but significant time periods between six months and a year continuously popped up as the 'least worst' correlation coefficients. Dosefactors of 0.5 or 1 seem to be favored: either the factor time is more important than concentration ($n = 0.5$), or they are about equal in importance ($n = 1$).



(a) Full time period, $n = 1$ (b) Full time period, $n = 1$



(c) Full time period, $n = 2$ (d) Partial time period, $n = 1$



(e) Full time period, $n = 0.5$ (f) Partial time period with time lag, $n = 1$

Figure 6.4: Illustrations of some of the options calculated in Appendix F.

ID	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*
CR1	1	1E+10	0.0677	0.592	-0.18	0.307	2	1E+10	0.0465	0.713	-0.2	0.287	0.5	1E+10	0.0835	0.508	-0.16	0.321
CR1	1	**52**	-0.133	0.292	-0.36	0.115	2	**52**	-0.112	0.376	-0.35	0.136	0.5	52	-0.094	0.458	-0.33	0.154
CR1	1	24	-0.101	0.422	-0.34	0.146	2	24	-0.062	0.622	-0.3	0.184	0.5	**24**	-0.129	0.304	-0.36	0.118
CR1	1	12	-0.041	0.746	-0.28	0.205	2	12	-0.011	0.93	-0.25	0.233	0.5	12	-0.075	0.552	-0.31	0.172
CR1	1	4	-0.018	0.887	-0.26	0.227	2	4	-0.004	0.977	-0.25	0.24	0.5	4	-0.043	0.733	-0.28	0.203
CR1	1	2	0.0059	0.963	-0.24	0.249	2	2	0.001	0.993	-0.24	0.245	0.5	2	-0.018	0.887	-0.26	0.227
CR1	1	1	0.0439	0.728	-0.2	0.285	2	1	0.0169	0.894	-0.23	0.26	0.5	1	0.0184	0.884	-0.23	0.261
CR2	1	1E+10	-0.129	0.307	-0.36	0.119	2	1E+10	-0.146	0.246	-0.38	0.101	0.5	1E+10	-0.118	0.348	-0.35	0.129
CR2	1	**52**	-0.202	0.107	-0.42	0.045	2	**52**	-0.173	0.169	-0.4	0.074	0.5	**52**	-0.199	0.111	-0.42	0.047
CR2	1	24	-0.148	0.238	-0.38	0.099	2	24	-0.085	0.503	-0.32	0.163	0.5	24	-0.18	0.151	-0.41	0.066
CR2	1	12	-0.052	0.681	-0.29	0.194	2	12	-0.004	0.976	-0.25	0.24	0.5	12	-0.093	0.462	-0.33	0.154
CR2	1	4	-0.018	0.886	-0.26	0.227	2	4	0.0205	0.871	-0.22	0.263	0.5	4	-0.048	0.704	-0.29	0.198
CR2	1	2	-0.008	0.95	-0.25	0.236	2	2	0.0292	0.817	-0.22	0.271	0.5	2	-0.034	0.786	-0.28	0.211
CR2	1	1	0.0124	0.922	-0.23	0.256	2	1	0.02	0.874	-0.23	0.263	0.5	1	0.0072	0.955	-0.24	0.251

Table 6.1: Correlation coefficient R (with confidence limits Rlo and Rup) and null hypothesis p for various dosefactors n and timeperiods *Period*. The timeperiod is given in weeks; if 1E+10 is given it means the entire previous period. **No time lag**. Emphasized values indicate highest correlation and lowest p per dosefactor n.

6.3 Model application

Opposite to the suggested probabilistic approach with both exposure modeling, the data which was generously provided by DHI Singapore contains hindcasted exposure modeling, which is in this report considered as deterministic input because of the supposed accuracy. As both the ordered response model and GUTS need to be first calibrated, and preferably followed by a validation round, only calibration is possible for these two models (considering that these are the only datasets available at time of writing). However, calibrating on weakly correlated data (Section 6.2) is not a recipe for accurately calibrated models. Furthermore, the measure of effect (coral cover) has besides the observed coral cover compatibility problems with both GUTS and the ordered response model:

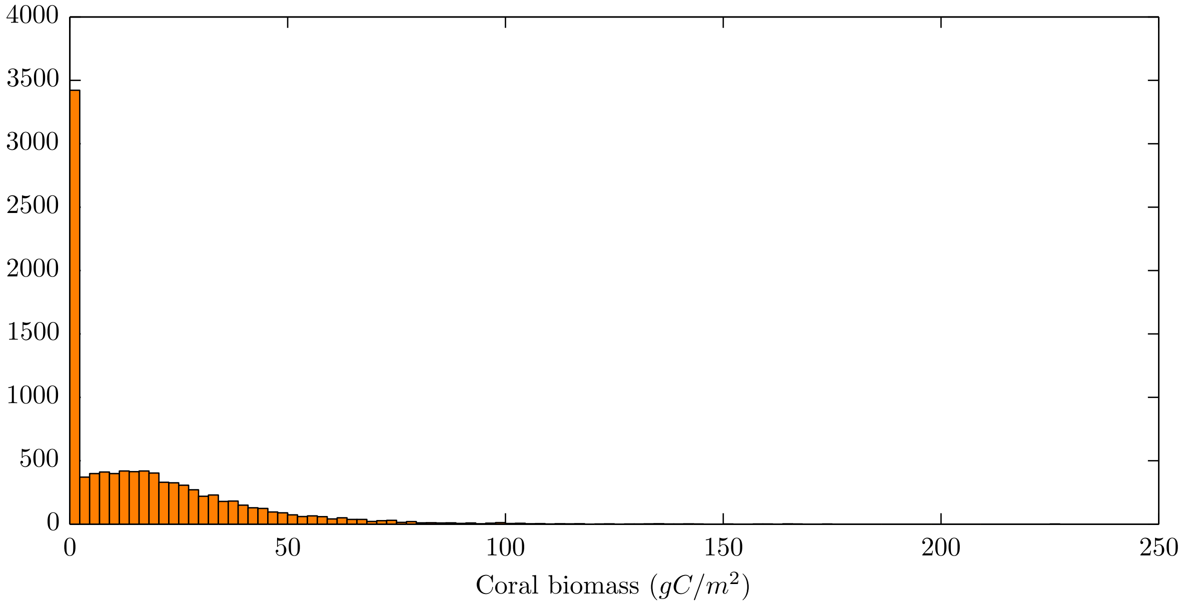
1. For the ordered response model, data is required which describes effects (divided in classes as described in Subsection 4.2.1) subjected to described exposures. If these effects are reviewed per individual, the *chance* for an effect class to occur equals the *portion* of the group of individuals reviewed which exhibit the effect from the effect class. If effects are only noted per group, the quantification is lost and only a *chance of occurrence* of the effect is given.

2. The ordered response model expects *end points*; effects after the exposures have happened.

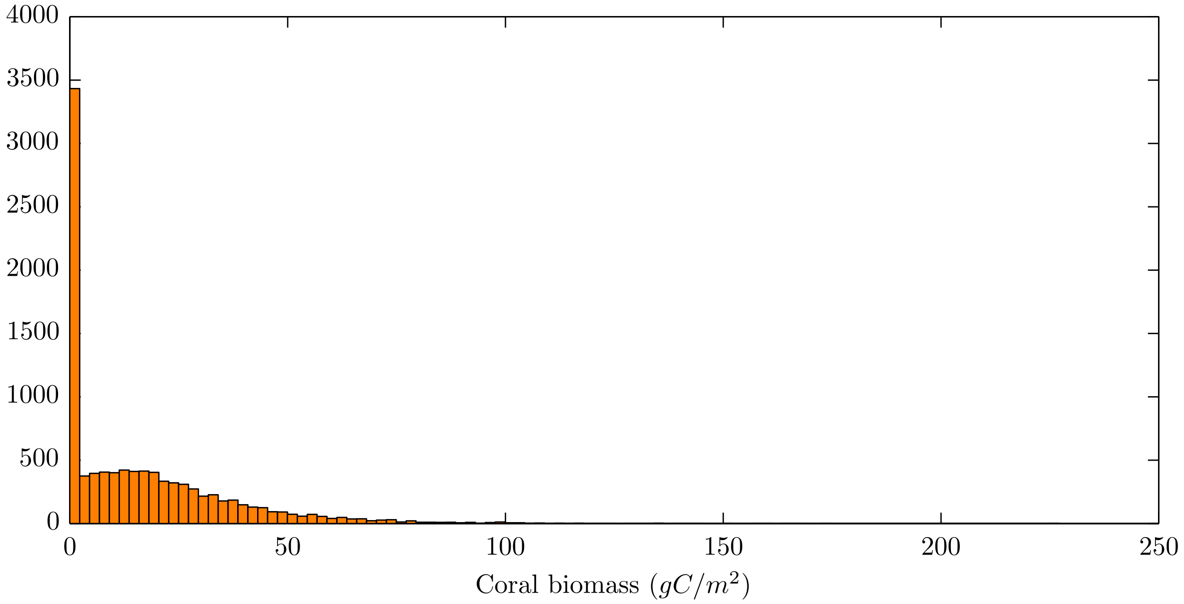
3. The GUTS model expects *timerecords* of *individuals* exposed to an exposure; as opposed to ordered response model for which endpoints are sufficient.

During preliminary investigations of the applicability of coral cover for GUTS and the ordered response model, no viable solution was found for the GUTS model: converting percentages to individuals cannot be done without more information. This made a GUTS application no longer viable. The ordered response model was easier to massage into coral cover percentages; but the effect classes had to be changed. Because coral cover measurements were available a half year before the modeling of the dredging induced excess TSS, this period can be assumed as the background variation. Any (negative) significant change outside this background variation could be possibly linked to the dredging exposure. Significant was defined here as outside the 95% confidence interval, but almost no points were found to be outside this interval; making application of the ordered response model no longer viable as well. In fact, reasonable results could only be found if the interval was changes to anything outside the top 50%; which does not make much sense to quantify.

Opposed to the GUTS and probit model is the coral productivity model. Because most parameters have a direct link to a biological background, viable estimates were found in literature in Larsen (2012). This made application much easier; changes in biomass are also easier to relate to coral cover changes than to the effect classes as mentioned in Subsection 4.2.1. However, as was alluded in Section 4.4, the high number of variables gave raise to concerns regarding the uncertainty of the final answer. The qualification 'high' is used here in comparison to the GUTS or ordered response model; in comparison to other energy budget models the number of variables could actually be qualified as 'low' (because it is a relatively simple energy budget model). To address the concerns of the relative high number of variables, a Monte Carlo simulation was made using the TSS signal of CR1 (modeling the entire period as described by the coral cover measurements) for the coral productivity model in Appendix C, Section C.6. The distribution of the findings are repeated here, together with another Monte Carlo simulation that includes the TSS signal on top of the assumed TSS background signal. Details of the Monte Carlo simulation can be found in Section C.6; histograms of the uncertainty regarding the final answer (with and without dredging) can be found in Figure 6.5.



(a) Coral biomass histogram without excess dredging induced TSS. Average biomass is 16.76.



(b) Coral biomass histogram with excess dredging induced TSS. Average biomass is 16.81.

Figure 6.5: Monte Carlo coral biomass distribution for the coral productivity model.

Figure 6.5 shows that the uncertainty regarding the final answer is very large, and also that the distributions of with and without dredging are almost the same; indicating no significant influence (which agrees with the findings of weak to no correlation in Section 6.2). The large uncertainty regarding the final answer is the product of a lot of variables in the coral productivity model having significant variation. The variation is much larger than what would be expected in nature; especially if these numbers are compared with the coral cover variation in the previous section. An attempt was made to reduce some of the uncertainty and/or better quantify the variability of light related parameters in Appendix D. However the largest uncertainty (knowledge wise) lies in the processes that describe the symbiont relationship between zooxanthellae en coral. Based on Appendix C, the following steps could be taken to possibly reduce uncertainty:

- Part of the uncertainty can possibly be solved by obtaining more accurate estimates of each variable; the majority of the estimates made in Appendix C are qualified as 'first-order estimates' and are based on Larsen (2012), which in turn often had to rely on literature to provide estimates. Experiments focused on understanding a single or a few parameters could lead to better and more precise estimates.

- Because seagrass also utilizes photosynthesis, but lacks the symbiont relationship of coral-zooxanthellae, the coral productivity model could also be used for seagrass. As the symbioses proved to be one of the more complex characteristics to capture in the coral productivity model by Larsen (2012), it could potentially reduce uncertainty significantly. However, it is very likely that seagrass has its own complexities. As the model has not been applied to seagrass in this document, this remains a preliminary recommendation and no definitive statement can be (or is) made.

- Depth of the coral is currently modeled as a stochastic variable. This could be taken out and replaced by a more detailed description of an applicable depth gradient.

- The relations describing irradiation are at the core of the productivity model, and irradiance is directly influenced by TSS concentrations. Reducing uncertainty here would significantly improve the model and help application for seagrass as well; calibration to local conditions would be another example of possibly significantly reducing uncertainty.

- Background exposures, where prominent examples are TSS and temperature, need to to be included as well. Especially coral is considered to be sensitive to temperature changes (specifically temperature anomalies, see also Appendix E). But even when temperature stays within 'natural' bounds, it can significantly influence biomass growth of coral. The local situation and forecast period of the required predictions can determine if this actually has significantly influence; for example in the Singapore Strait water temperature is considered to be stable within a range and average of respectively three degrees and 29.7°C (Gin et al., 2000; Larsen, 2012). In this case, this was assumed to have no significant influence on production (Gin et al., 2000; Larsen, 2012).

It is stressed that this is not a disqualification for the relations used to describe coral/-zooxanthellae in the coral productivity model, it is only a description of the accuracy of the obtained final answer. What seems to be (acceptable) natural variability in each separate parameter, does not seem to lead to a realistic variation in the final answer (when globally compared to the variability of coral cover in CR1 in Figure 6.1). The list of items mentioned above could help in reducing variability; obtaining more accurate estimates of variables would be a good starting point as the estimates made do not always have a desirable accuracy. After this, the model can be re-assessed and, if the variation is more likely, to see if the used relations are indeed an accurate description.

Chapter 7

Conclusion

7.1 Conclusions

The conclusions are given as answers to the research questions posed in Section 1.2.1. The research goal was the development and application of a risk-based approach to assess effects of temporary exposures on sensitive receivers due to dredge plumes.

I How do dredging related activities (sources) contribute to exposure signals, and how can these be quantified?

According to used literature, for a location sufficiently far from the dredging operation, exposures primarily develop from fine sediments which are released by the dredging equipment. The relevant processes are different per equipment type and the particular utilization of the equipment type. Fine sediment content is part of the dredged material, of which a fraction is released by the dredger during the dredging operation, and is considered to be a primary contributor to exposure signals because of fine sediment being able to stay suspended in the water column for long periods of time.

II How do exposure signals stemming from dredging activities develop, and how can these be quantified?

The sum of fine sediment sources released by the dredging equipment will most likely lead to a dynamic plume, which consists of interacting, complex three-dimensional fluid motions: sediment characteristics, bathymetry and dredging equipment type are just a few of the influencing factors. Ongoing research is done towards a proper schematization, but currently the dynamic phase is captured in conservative estimates.

These estimates serve as a source term for the passive plume phase. During the passive plume phase, advection and diffusion are dominant processes which are well understood processes. Assessing these processes as well understood is common practice[1]. MIKE 21/3 or Delft3D, both implementing hydrodynamic and transport models, can be used to schematize the passive plume. Modeling of a dredge plume was not undertaken in this document as results of a dredge plume model (hindcasted and calibrated to local measurements) was made available.

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[1] Personal communication with Dr. ir. M. van Koningsveld

III Which aspects of exposure signals are relevant for quantifying exposures for sensitive receivers?

Once a passive plume reaches a sensitive receiver, the two dominant exposures in literature are considered to be sedimentation and fine suspended sediment concentrations. While these are purely dredging induced exposures, several other exposures can occur near the sensitive receiver which can be as important as dredging exposures; examples include temperature or changes in suspended background matter. In general, the shape of the exposure signal contains relevant aspects of temporal variable exposure signals: intensity, duration and persistence. For seagrass and corals, suspended sediment can be seen as an energy limiting exposure as it attenuates light which in turn reduces photosynthetic capabilities (considered to be the primary energy source for corals and seagrass).

IV How can effects, concerning sensitive receivers, be quantified?

Which range of exposure signals and to what extent these can influence a sensitive receiver depends very much on the type of receiver and the current (and past) conditions it was exposed to. Specifically in the case of coral, temperature is an example of an exposure which is not influenced by dredging, but where coral is known to be sensitive for. Exposure characteristics such as intensity, duration and persistence, but also receiver characteristics such as the species it belongs to, or current (and past) conditions it was exposed to all determine possible effects.

On an abstract level, effects are mere discrete points of health or well being. Ideally, a gradient would be given which describes everything between mortality and perfect health. As this is difficult to do, a number of discrete points in health, effect classes, were given: growth/positive effects, physiological changes, disease or temporary damage, and mortality.

These effect classes can be modeled using models originally found in toxicology, even though suspended sediment or sedimentation is not considered toxic. These toxicological models focus on one or more exposures, and relate only these exposures to effects. Especially for relatively high levels of these exposures, where these exposures can be considered dominant and perhaps even somewhat ‘toxic’, this seems to be a good approach. Examples of such toxicological models are the General Unified Threshold Model (GUTS) and the ordered response model.

On the other hand, if there are a multitude of exposures which all have a significant chance to induce effects in the sensitive receiver, an energy budget approach seems more fitting. In an energy budget approach, relevant processes are modeled which together represent biomass or size. This focus on growth, or lack of growth, makes energy budget approaches also applicable in case there are no significant negative exposures expected. Growth, though technically possible, is awkward to achieve in toxicological models because of the assumed relation between exposure and effect: higher exposures lead to a higher chance of the effect occurring. The energy budget approach is not without flaws either: an implementation usually consists of many parameters, each with their own uncertainty. If a portion of these parameters cannot be accurately estimated, the final answer will see great variability; to a point where the parameter uncertainty overrules any other (expected) uncertainty.

Mangroves, mentioned as part of three sensitive receivers, are not as ‘sensitive’ as coral or seagrass. In fact, mangroves are often encountered in systems where coral and/or seagrass have to be regarded as well; from practical experience and regular occurring conditions for said mangroves corals and/or seagrass are the bottleneck regarding effects.

V How can data concerning exposure and effect be gathered and applied (e.g. in models) for sensitive receivers?

Specifically for the ordered probit, GUTS and coral productivity model it can be said that reliable data is lacking. An attempt was made to use field measurements originally intended for management purposes, but this proved to be unsuccessful. Unsuccessful is defined here as successful calibration and/or validation of the three earlier mentioned models. Because this was field data belonging to an existing project, exposures were relatively low (the project had to, and probably did, meet its criteria); more interesting would be higher intensity exposures, as these will lead to more severe effects: these effects need to be encountered to 'know where the limit is'. More, detailed data is required in order to properly calibrate and validate the earlier proposed models; to minimize impact on existing communities these tests could best be executed in laboratories or artificial, small colonies.

VI How can a risk-based approach be applied to current dredging practice?

A risk-based approach gives insight in uncertainties, and uncertainties are plenty in dredging practice and ecological modeling. Insight in these uncertainties can lead to early recognition of weak spots, which in turn help managing risks. Understanding the risks related to uncertainties can also be helpful in determining the value of an ecosystem (or damage to this).

7.2 Recommendations

Dredging exposure experiments

As was mentioned in the conclusions, data regarding ecological modeling with emphasis on dredging exposures is lacking. Experiments in basically all facets of ecological modeling are needed before any of the above mentioned models can be applied. These models need to be properly calibrated and validated first for a range of exposures before any can be used in practice. In the end, if one or more models get validated, this could lead to a better, more optimized way of predicting (and understanding!) effects due to exposures. Attention should probably first go out to the simpler toxicological models as they are simpler models. The energy budget principle is a more generic, holistic approach which will presumably take longer to get properly implemented, but in the end describes health better than the toxicological models.

Background exposures and conditions

Just as important as the dredging induced exposures, is insight in the occurring background exposures. Dynamics and properties of these are often oversimplified or even neglected, which could lead to effects of dredging being exaggerated or neglected: both are far from ideal. A simple example can be found in so called 'algal blooms', sudden increases in algae in the water, which attenuate light, which in turn reduce photosynthetic capacity of the zooxanthellae living in symbiosis with the coral, which ultimately can have a significant impact on coral cover.

Community dynamics

Attention in this document mainly went out to individuals, and groups of individuals belonging to the same species. However, it is common to encounter different species which form a community. These community dynamics have been treated qualitatively with occasionally a reference to an existing model, but it would be my recommendation to properly include these in the modeling train as well. Small decreases in species population could lead to dramatic changes in community dynamics (phase shifts or even loss of an entire community). A possible approach would be trying to identify key species: species which are invaluable to a community and, for example if removed, can even lead to the collapse of an entire community.

Valuation of (damage to) ecosystem services

If effects can be accurately modeled, these effects still need to deemed acceptable: in other words, do the (envisioned) benefits of the dredging operation exceed the (envisioned) risks of the effects. One way to do this is to by attributing monetary value to the sensitive receivers, and the influence of these effects on this value. However, attributing value to ecosystem services is a complex and controversial exercise and should be done carefully. Sectors which might also be facing similar problems include the oil and nuclear industry, and their experience could be used as well.

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Appendix A

Ordered response model

A commonly used method in toxicology to describe exposure-effect relationships is the probit (or logit) model. (In toxicology these are called dose response relationships, but in the following these are called exposure-effect relationships.) The foundations of this model is the assumption of a relation between exposure effect is shaped like a cumulative normal distribution (or cumulative logistic distribution for logit). in its basic form, given certain exposure(s), only a binary response is given: effect is present with chance X, or it is not present with chance $1 - X$.

It should be noted that exposure (or dose) is not limited to a single type of exposure; multiple exposures can be combined into a representative exposure. Effect does not have to be binary either: when an ordered response model is used, an arbitrary number of responses can be defined (for instance linking the responses to the effect classes of Subsection 4.2.1). Originally proposed by Finney (1971) for toxicological modeling, and proposed for ecological modeling in Becker (2011), the following is an excerpt of the description in Becker (2011).

Please note: *Originally proposed by Finney (1971) for toxicological modeling, and proposed for ecological modeling in Becker (2011), the following can be considered a direct citation of chapter 2 of Becker (2011) as the primary source. For brevity and readability, quotation marks have been left out.*

A.1 Governing equations

The ordered response model can be explained by first defining ordered alternatives from 1 to m; these values have no real meaning, but are index values. For individual i the values of k explanatory variables are denoted by the $k \times 1$ vector x_i and the outcome of the dependent variable is denoted by y_i. The outcome y_i is not used directly, but related to an index function (Becker, 2011):

$$y_i^* = x_i'\beta + \varepsilon_i, \quad \varepsilon_i \sim \mathrm{N}(0, 1). \qquad [A.1]$$

Here β is a $k \times 1$ vector of unknown parameters and ε_i is the unobserved disturbance for individual i. The outcome y_i is the observed variable and is related to the index function by $(m - 1)$ threshold values $\tau_1 < \tau_2 < \cdots < \tau_{m-1}$ in the following way (Becker, 2011):

$$\begin{aligned} y_i &= 1 && \text{if} \quad -\infty < y_i^* \leq \tau_1, \\ y_i &= j && \text{if} \quad \tau_{j-1} < y_i^* \leq \tau_j,\ j = 2, \ldots, m-1, \\ y_i &= m && \text{if} \quad \tau_{m-1} < y_i^* < \infty. \end{aligned} \qquad \text{[A.2]}$$

In the index function [A.1], $x_i'\beta$ represents the systematic preference and ε_i the individual-specific effect. When Φ (the normal cumulative distribution function) is the CDF of ε_i, then (Becker, 2011):

$$\begin{aligned} p_{ij} = \Pr(y_i = j) = \Pr(\tau_{j-1} < y_i^* \leq \tau_j) &= \Pr(y_i^* \leq \tau_j) - \Pr(y_i^* \leq \tau_{j-1}) \\ &= \Phi(\tau_j - x_i'\beta) - \Phi(\tau_{j-1} - x_i'\beta), \quad j = 1, \ldots, m, \end{aligned} \qquad \text{[A.3]}$$

Where, only here, $\tau_0 = -\infty$ and $\tau_m = \infty$. The parameters that have to be estimated are β and the $(m-1)$ threshold values. The explanatory variable x_i should not contain a constant term, otherwise the threshold parameters are not identified (Becker, 2011).

A.2 Application

In Becker (2011), a practical application was given for fish responses to suspended sediment, where the representative exposure consisted of both suspended sediment concentration (C, mg/L) and duration of exposure (T, h) and the response was either behavioral effects, sub-lethal effects or lethal effects (indexed to 1, 2 and 3; $m = 3$). Data was fitted with an ordered probit model, as described in Equation A.1 (Becker, 2011):

$$\text{SEV}_i^* = \beta_1 \log(C_i) + \beta_2 \log(T_i) + \varepsilon_i, \quad i = 1, \ldots, 171, \qquad \text{[A.4]}$$

None of the data points showed a nil effect (which was effect class 0), which resulted in the discarding of this category. A scatter plot of the data on the grouped SEV scale is shown in Figure A.1a (Becker, 2011).

Fitting of the ordered probit model was done using Maximum Likelihood; cumulative probabilities are shown in Figure A.1b which where estimated using (Becker, 2011):

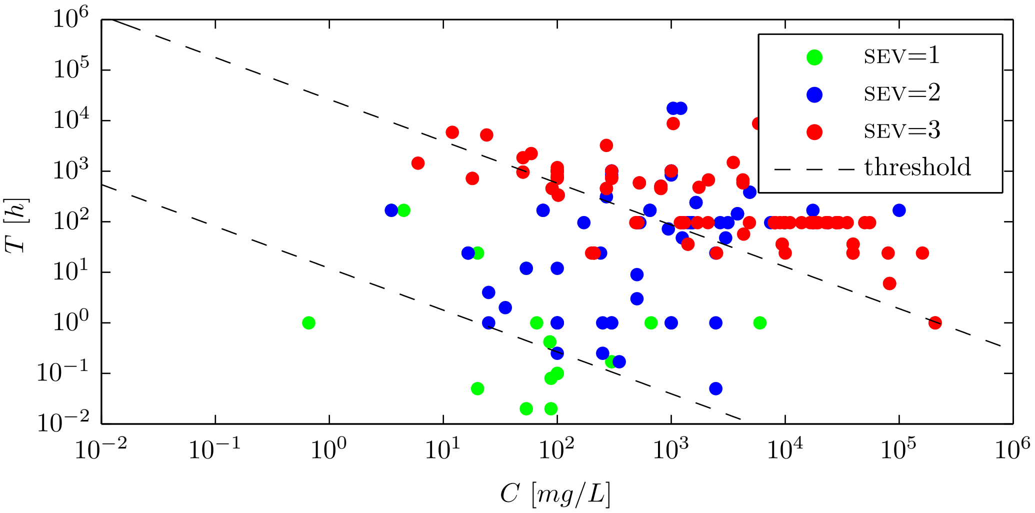
$$\Pr(\text{SEV} \leq j) = \Phi(\hat{\tau}_j - x'\hat{\beta}), \qquad \text{[A.5]}$$

where $x_1 = \log(C)$, $x_2 = \log(T)$, $\hat{\tau}_j$ is the estimate for the jth threshold value and $\hat{\beta}$ is the estimate for the parameter vector β in the probit model. A special case arises when $j = m$, since $\Pr(\text{SEV} \leq m) = 1$ for every value of SEV^*.

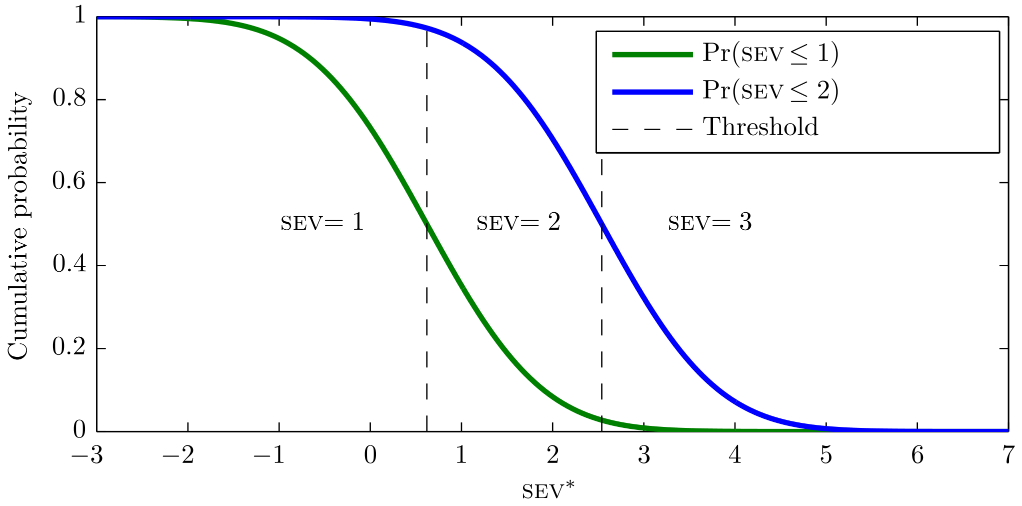
A.3 Variable exposure

In the previous case of concentration-duration as exposure, only constant concentrations were used. If the model has been fitted to (for example laboratory) tests with constant concentrations, newly found variable concentrations can be used to predict effects by rewriting the model in Equation A.4 into (Becker, 2011):

$$\text{SEV}^* = \beta_2 \log(C^n\ T), \qquad n = \beta_1/\beta_2, \qquad \text{[A.6]}$$



(a) Scatter plot on a log-log scale with severity of ill effect (SEV) as a function of suspended sediment concentration C and exposure duration T. SEV $= 1$ leads to behavioral effects, SEV $= 2$ to sublethal effects and SEV $= 3$ to lethal and paralethal effects. The threshold values are represented by the two black lines. Description and image courtesy of Becker (2011).



(b) Exposure-effect curves after fitting the ordered probit model. The cumulative probabilities Pr(SEV$\leq j$) are plotted as a function of effect class SEV*. The probability Pr(SEV≤ 3) is 1 for every value of the representative exposure $x'\hat{\beta}$; threshold values are represented by the two black lines. Description and image courtesy of Becker (2011).

Figure A.1: Data application of the ordered response model done in Becker (2011).

where the term $C^n\ T$ may be considered as a dose factor (Ten Berge et al., 1986). This factor may be expressed as (Becker, 2011):

$$\int [C(t)]^n \, \mathrm{d}t, \tag{A.7}$$

The values for β_1 and β_2 have already been established in the effects assessment; the value of SEV can then be determined (Becker, 2011). Incidentally, if $\beta_1 \approx \beta_2$ Haber's rule is found (Ten Berge et al., 1986); Haber's rule states that $C \times T = \mathit{constant}$ (Becker, 2011).

Appendix B

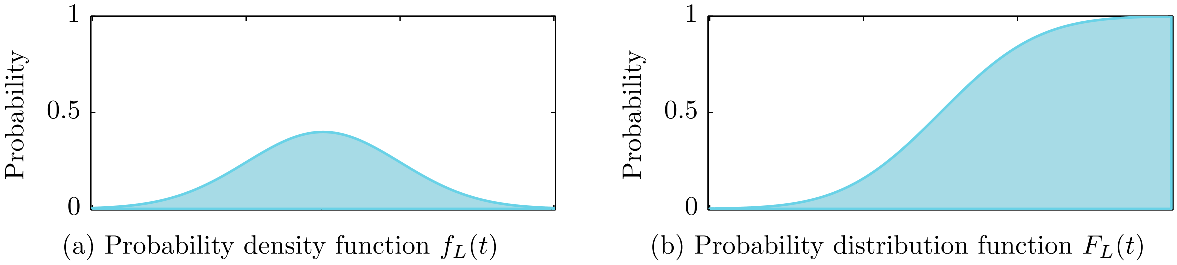
General Unified Threshold Model of Survival

A simplified model of the DEB theory, which only models survival, is the General Unified Threshold Model of Survival (GUTS) (Jager et al., 2011). This is actually a framework that can take the form of a number of Toxicokinetic-toxidynamic (TKTD) models. Jager et al. (2011) described TKTD models as simulating the time-course of processes leading to toxic effects on organisms. This is done by translating an external concentration of a dose to an internal concentration over time. These internal concentration are then qualitatively linked to the effect at the level of the individual organism over time. All models in GUTS share the fact that a simple one-compartment model with first-order kinetics is used in order to approximate the uptake and excretion of a dose.

B.1 Derivation survival function

The probability of an object to have failed at time T can be described as the chance that it failed between $t = 0$ and $t = T$. This is described formally in the probability distribution function of failure, $F_L(t)$. Depending on the subject and context of modeling, 'failure' could be replaced by 'death' as a more suitable description.

The probability density function $f_L(t)$ is the derivative of the probability distribution function $F_L(t)$; or, in other words, the probability distribution $F_L(t)$ is the cumulative of the probability density function $f_L(t)$. The difference between $f_L(t)$ and $F_L(t)$ is shown in Figure B.1 for a normal distribution.



(a) Probability density function $f_L(t)$ (b) Probability distribution function $F_L(t)$

Figure B.1: Graphical representations of $f_L(t)$ and $F_L(t)$. Shown here is an *(arbitrarily chosen!)* normal distribution.

The proportionality of the probability of failure can be expresses as a failure rate (Vrijling et al., 1997). The failure rate, or hazard rate, is defined as $r(t)$.

$$r(t) = \frac{f_L(t)}{1 - F_L(t)}$$

As $f_L(t)$ is the derivative of $F_L(t)$, this can be used to replace $f_L(t)$.

$$\begin{aligned} f_L(t) &= \frac{dF_L(t)}{dt} \\ \frac{dF_L(t)}{dt} &= (1 - F_L\,(t))\,r(t) \end{aligned}$$

This is a differential equation, which has the following solution for $F_L(t)$ (Vrijling et al., 1997):

$$F_L(t) = 1 - exp\int_0^t -r(\tau)d\tau \tag{B.1}$$

In the above equation the variable τ is the integration variable; this has been done to emphasize the difference between the integration variable τ from the upper bound t of the integral. The probability of non-failure or survival is the complement of the probability of failure. The distribution function of survival $S_L(t)$ is therefore defined as $1 - F_L(t)$:

$$S_L(t) = 1 - \left(1 - exp\int_0^t -r(\tau)d\tau\right) = exp\int_0^t -r(\tau)d\tau \tag{B.2}$$

In order to approximate the hazard rate, a simple one-compartment model with first-order kinetics can be used (Kooijman and Bedaux, 1996; Jager et al., 2011). The one-compartment model is one of the simplest models which still is realistic from a biological point of view (Kooijman and Bedaux, 1996). This can best be compared to a bathtub, which is filled and drained at the same time. If the filling exceeds the draining for a certain amount of time, the bathtub will overflow; in the analogy used here this means a chance to 'fail' or 'die'.

$$\frac{dC_i(t)}{dt} = k_a C_e(t) - k_e C_i(t)$$

In this one-compartment model, the internal concentration (in an organism) of the dose is represented by $C_i(t)$, the external concentration by $C_e(t)$, and the constants representing accumulation and elimination are respectively k_a and k_e. For comparison with an external concentration however, a scaled internal concentration (which can be compared directly to an external concentration) is of more use. This scaling can be done by dividing both sides of the above equation by k_a/k_e (Jager et al., 2011). This scaling has the additional benefit of removing the need to find the accumulation rate constant k_a (survival data alone can never provide information about the accumulation rate constant (Jager et al., 2011)); this factor is readily included in the scaled internal concentration $C_i^*(t)$. Furthermore, because of this scaling, external concentrations can be directly linked to internal concentrations, removing the need to measure actual internal concentrations.

$$\begin{aligned}
C_i^*(t) &= \frac{k_e}{k_a} C_i(t) \\
\frac{dC_i(t)}{dt} &= k_a C_e(t) - k_e C_i(t) \\
\frac{k_a}{k_e} \frac{dC_i^*(t)}{dt} &= k_a C_e(t) - \frac{k_e k_a}{k_e} C_i^*(t) = k_a C_e(t) - k_a C_i^*(t) \\
\frac{dC_i^*(t)}{dt} &= k_e \left(C_e(t) - C_i^*(t)\right)
\end{aligned}$$

A next step could be to link internal concentrations to scaled damage, which is proportional to the actual (but unknown) damage level and has similar units as the internal concentration. A scaled damage level is used because actual damage cannot be measured; the rescaling is analogous to the scaling method used for internal concentrations.

$$\frac{dD_i^*(t)}{dt} = k_r \left(C_i(t) - D^*(t)\right)$$

The addition of scaled damage can only slow down the kinetics of the dose metric; the upper limit would be set by the elimination rate from the measured body residues (Jager et al., 2011). But, according to Jager et al. (2011), without actual measured body residues to estimate a relevant elimination rate constant k_e (or read-across from related chemicals or quantitative structure activity relationships (QSAR)) it is generally not a good idea to use a state of damage. Relevant data from sensitive receivers is already hard to come by at this moment, expecting to find measured body residues data is therefore not feasible. This means the only viable option is to use the scaled internal concentration. Linking damage to scaled internal concentrations can be done by setting k_r to infinitely fast (k_r to ∞). Consequences of using only an internal scaled concentrations model to predict damage:

- Either the elimination of the toxicant k_e is assumed to be linked directly to body residues;
- Or the damage repair rate k_r dominates the elimination rate k_e ($k_r >> k_e$).

It is however entirely possible that the elimination rate k_e dominates the repair rate k_r ($k_r << k_e$). *Mathematically*, this does not need to have dire consequences if only a scaled internal concentration model is used; the *physical* meaning of the parameter k_e is however lost. Instead, it represents a “dominant” rate (either k_e, k_r or a mix between the two). For simplicity's sake, k_e will still be used in any further derivation using a scaled internal concentration. Despite the need for more detailed data, scaled damage could still be a worthwhile addition because it makes physical interpretation clearer; it separates the toxicokinetic (TK, “uptake/removal”) from the toxidynamic (TD, “damage relation”) part and does not have an ambiguous k_e parameter. However, as stated before, given the current scarcity of available data for sensitive receivers, scaled damage cannot be easily derived and a scaled internal concentration model is preferred from a practical point of view.

The differential equation with a scaled internal concentration is a first order differential equation, which can be solved using the method of integrating factors.

$$\begin{aligned}\frac{dC_i^*(t)}{dt} &= k_e\left(C_e(t) - C_i^*(t)\right)\\ \mu(t) &= e^{k_e t}\end{aligned}$$

Multiplying both sides with the integrating factor $\mu(t)$:

$$\mu(t)\frac{dC_i^*(t)}{dt} + \mu(t)k_eC_i^*(t) = \mu(t)k_eC_e(t)$$

The product rule can be used to replace the two terms on the left hand side. The product rule states that $(\mu C)' = \mu C' + \mu' C$; the second term on the left hand side contains $\mu(t)k_e$ which is the derivative of $\mu(t)$. Rewriting the left hand side using the product rule leads to the solution of the differential equation:

$$\begin{aligned}\frac{d}{dt}\left[\mu(t)C_i^*(t)\right] &= \mu(t)k_eC_e(t)\\ \mu(t)C_i^*(t) &= \int \mu(t)k_eC_e(t)dt + K\\ C_i^*(t) &= k_e e^{-k_e t}\int e^{k_e t}C_e(t)dt + e^{-k_e t}K\end{aligned}$$

The integration constant K can be found by imposing that $C_i^*(0)$ is the scaled internal concentration at $t = 0$:

$$\begin{aligned}C_i^*(0) &= k_e e^{-k_e\cdot 0}\int e^{k_e\cdot 0}C_e(0)dt + e^{-k_e\cdot 0}K = 0 + K = K\\ C_i^*(t) &= k_e e^{-k_e t}\int e^{k_e t}C_e(t)dt + e^{-k_e t}C_i^*(0)\end{aligned}$$

$C_i^*(0)$ is assumed to be negligibly small (Kooijman and Bedaux, 1996). This needs to be validated every time this derivation is used.

$$C_i^*(t) = k_e e^{-k_e t}\int_0^t e^{k_e\tau}C_e(\tau)d\tau \qquad [B.3]$$

The hazard rate is assumed to be linearly increasing with the internal concentration of [B.3] above a certain threshold z. Including a background hazard rate $r_b(t)$ and a proportionality constant k_k (the 'killing rate') leads to the following definition for a hazard rate of an individual organism (Kooijman and Bedaux, 1996; Jager et al., 2011).

$$r(t) = \begin{cases} r_b(t) & \text{if } C_i^*(t) - z \leq 0\\ r_b(t) + k_k\cdot\left(C_i^*(t) - z\right) & \text{Otherwise}\end{cases} \qquad [B.4]$$

The background hazard rate can be explained as the hazard rate in a control group; a group that is not exposed to a dose. Assumed is that effects in a control group are stochastic events, independent from effects caused by the dose in the other group; therefore the two hazard rates can be summed Jager et al. (2011). For relatively short time scales, the background hazard rate could be assumed to be constant. For relatively long timescales, background hazard rates are almost certainly variable.

If the first case of [B.4] is ignored, which has a rather trivial solution, the second case of the hazard rate of an individual organism can be expanded in conjunction with [B.1], [B.2] and [B.3] to define individual failure and survival distributions respectively. Only the individual survival distribution has been expanded below, because failure distributions can be easily derived from survival distributions (and vice versa, see [B.2]).

$$\begin{aligned}
S_i(t) &= exp\int_0^t -r(\tau)d\tau \\
S_i(t) &= exp\left[-\int_0^t r_b(\tau) + k_k \cdot \left(C_i^*(\tau) - z\right) d\tau\right] \\
S_i(t) &= exp\left[-\int_0^t r_b(\tau)d\tau\right] \cdot exp\left[-k_k \cdot \int_0^t \left(C_i^*(\tau) - z\right) d\tau\right] \\
S_i(t) &= exp\left[-\int_0^t r_b(\tau)d\tau\right] \cdot exp\left[-k_k \cdot \int_0^t \left(k_e e^{-k_e\tau} \cdot \int_0^t e^{k_e\sigma} C_e(\sigma)d\sigma - z\right) d\tau\right]
\end{aligned} \tag*{[B.5]}$$

The integration variables τ and σ both represent time ranging from 0 to t; t can't be used because it is already present in the upper bound of both integrals. The first case of [B.4] can be found by removing the second exponential. For the simplified case of a constant background hazard rate and a constant external concentration the expansion of [B.4] results in the following expression:

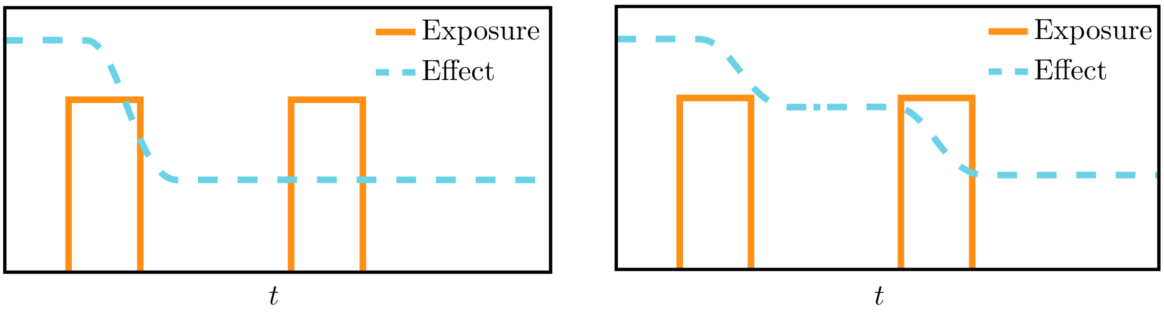
$$S_i(t) = \begin{cases} exp\left[-r_b t\right] & \text{if } C_i^*(t) - z \leq 0 \\ exp\left[-r_b t\right] \cdot exp\left[\frac{-k_k}{k_e}\left(-C_e + C_e k_e t + C_e e^{-k_e t} - z k_e t\right)\right] & \text{Otherwise} \end{cases}$$

In order to go from an individual organism survival distribution to an arbitrary organism survival distribution, [B.5] has to be multiplied with the probability distribution of the threshold z, $f(z)$. After integration over z, $S_i(t)$ can be transformed from a survival distribution conditioned on threshold z, to an unconditional survival distribution $S_L(t)$ (Jager et al., 2011).

$$S_L(t) = \int_0^\infty S_i(t) f_z(z) dz \tag*{[B.6]}$$

B.1.1 Special cases

The equation in [B.6] is the general form of the GUTS framework. Two specific forms of this framework differ in their assumption of the cause of mortality. The first group of models assumes that individuals have different sensitivities ("Individual Tolerance, IT"). This is generally the only assumption mentioned in ecotoxicological textbooks to explain sigmoidal dose response curves. Challenging this assumption (amongst others by Kooijman and Bedaux (1996)), the second group of models assumes individuals are identical but have a certain amount of variability. Mortality is therefore treated as a stochastic process ("Stochastic Death, SD") (Jager et al., 2011). IT models can be derived from [B.6] by making k_k infinitely large, effectively implying failure once the threshold is exceeded. SD models can be found by using a Dirac delta distribution for $f_z(z)$; this reduces $S_L(t)$ to $S_i(t)$ as there is only a single, constant z value (Jager et al., 2011). A mixture between SD and IT is also possible. A simplified illustration of the difference between these two can be found in Figure B.2.



(a) IT: Stochastic distribution regarding resistance to effects. The first exposure has effect on the fraction that has not enough resistance. Subsequent similar exposures do not provoke any response.

(b) SD: Effect is related to the exceedance of the species' resistance. This means that subsequent similar exposures have similar effects like the first exposure.

Figure B.2: Example of two different response types for an effect on a (fictitious) population of organisms, both responding to the same intermittent exposure signal.

In the case of SD models and a constant concentration, if k_e is extremely small (approaching zero) a Weibull distribution with shape parameter 2 can be found; if k_e is extremely large (approaching infinity) an exponential distribution (or Weibull with shape parameter 1) can be found (Kooijman and Bedaux, 1996). A choice between SD or IT can be made for valid reasons, but a choice is not obligatory: if the full model is used, the found values for the parameters can be interpreted and judged if the behavior is SD, IT or anything in between.

B.2 Estimating parameters

Estimating the parameters z, k_k, k_e, and if required parameters from $h_b(t)$, of [B.6] is not straightforward. Recorded data of survival experiments usually contain counts of the number of individuals of a group that exhibit effects due to a dose, measured at fixed times. Because of this, the exact time individuals show a response is not known, only a period of time is known (Kooijman and Bedaux, 1996; Jager et al., 2011); this is also known as interval censoring. This means time intervals, and the probabilities accompanying them, should be used in calculations.

The outcome of a survival experiment can be seen as a trial from a multinomial distribution (Kooijman and Bedaux, 1996). A binomial distribution can be explained as a coin toss where heads indicates success with probability p; this means tails has a probability of $1-p$. A multinomial distribution is a generalization of a binomial distribution: an arbitrary number of outcomes are now possible. The probability function of a multinomial distribution is:

$$P\left(X_1=x_1,...,X_n=x_n\right)=\frac{N!}{\prod_{i=1}^{n}x_i!}\prod_{i=1}^{n}k_i^{x_i}$$
$$P\left(X_1=x_1,...,X_n=x_n\right)=N!\prod_{i=1}^{n}\frac{k_i^{x_i}}{x_i!}$$

Where k_i are constants with $k_i>0$ and x_i are non negative integers. Both parameters also need to abide to:

$$\sum_{i=1}^{n}x_i=N \qquad\qquad \sum_{i=1}^{n}k_i=1$$

Considering an experiment is setup in such a way that all individuals exhibit effects in one of the time intervals. If they didn't, any remaining individuals are assumed to exhibit effects between the last observation and $t=\infty$. If θ is a vector of parameters to be estimated, and $S_L(t)$ of [B.6] is used, the probability for a random individual to die between $t-1$ and t is $S_L(t_{i-1},\theta)-S_L(t_i,\theta)$ (together adding up to one). The number of survivors is denoted as $y=(y_0,\ldots,y_n)$, with $y_i=y(t_i)$ for the times $t_0=0<t_1<\ldots<t_n$ (Jager et al., 2011). Using this in the given multinomial distribution leads to the probability of obtaining the actual counts given the estimation (denoted by means of an overbar) of the survival function; this is considered to be a maximum likelihood estimate constructed using the survival function in [B.6] and experimentally acquired interval censored survivor data (Kooijman and Bedaux, 1996; Jager et al., 2011).

$$x_i=y_{i-1}-y_i$$
$$k_i=S_L(t_{i-1},\theta)-S_L(t_i,\theta)$$
$$\sum_{i=1}^{n}x_i=y_0$$
$$P\left(\overline{y}_i=y_i\right)=P\left(\overline{x}_i=x_i\right)$$
$$P\left(\overline{x}_i=x_i\right)=l(y|\theta)=y_0!\prod_{i=1}^{n+1}\frac{k_i^{x_i}}{x_i!}$$

Where $S_L(t_{i+1},\theta)$ and y_{n+1} are set to zero. When working with maximum likelihood, it is often computationally easier to use its logarithm. Taking the natural logarithm of the previous equation gives the log-likelihood (Jager et al., 2011):

$$l(y|\theta)=y_0!\prod_{i=1}^{n+1}\frac{k_i^{x_i}}{x_i!}=\frac{y_0!}{\prod_{i=1}^{n+1}x_i!}\prod_{i=1}^{n+1}k_i^{x_i}$$
$$\ln l(y|\theta)=\ln\left(\frac{y_0!}{\prod_{i=1}^{n+1}x_i!}\right)+\ln\left(\prod_{i=1}^{n+1}k_i^{x_i}\right)$$

The first term on the right hand side does not contain any of the parameters that need to be estimated, therefore this term is not important in obtaining a maximum likelihood and can be ignored:

$$\ln l(y|\theta) \propto \ln\left(\prod_{i=1}^{n+1} k_i^{x_i}\right) = \sum_{i=1}^{n+1} \ln\left(k_i^{x_i}\right) = \sum_{i=1}^{n+1} x_i \cdot \ln\left(k_i\right)$$

$$\ln l(y|\theta) \propto \sum_{i=1}^{n+1} (y_{i-1} - y_i) \cdot \ln\left(S_L(t_{i-1}, \theta) - S_L(t_i, \theta)\right)$$

If multiple time-records of concentrations are available, these can be used together to gain an overall likelihood by multiplying likelihoods (or by summing log-likelihoods). If there are a total of r time-records:

$$\ln l(y|\theta) \propto \sum_{i=1}^{r} \sum_{j=1}^{n+1} (y_{i,j-1} - y_{ij}) \cdot \ln\left(S_L(t_{i,j-1}, \theta) - S_L(t_{ij}, \theta)\right) \qquad \text{[B.7]}$$

Finding optimal values for this maximum likelihood function is not straightforward; the function has multiple degrees of freedom and local maxima. To make matters worse, the global maximum usually has a very steep slope, which occurs rather suddenly; algorithms that take steps to see if there is a gradient leading to a new maximum easily get stuck (meaning the solution is sensitive to the chosen start values, see Figure B.3 for an example). A way out of this is by using a brute force approach: A lot of (rough) optimization efforts with many different starting values. The most promising effort is then used in a more detailed optimization attempt.

B.3 Ordered response

Depending on the project, death/survival might not be the most relevant or the only effect to research; no effect/effect, behavioral effects or non-lethal effects are examples of other interesting quantal (binary) effects. The aforementioned effects are ordered: the effect progressively becomes ‘worse’. As the GUTS model assumes an organism to approximate a one-compartment model with effects occurring above a certain threshold, it seems reasonable to hypothesize that (for the same toxic substance) uptake, elimination and damage repair rates are constant: Only the threshold value should vary between various effect categories. If there are m response categories, this would lead to estimates for k_k, k_e, $\sum_{i=1}^{m} z_i$. Depending on the type of model used, the maximum likelihood equation [B.7] changes into:

$$\sum_{i=1}^{m} \sum_{j=1}^{r} \sum_{k=1}^{n+1} (y_{i,j,k-1} - y_{ijk}) \cdot \ln\left(S_L(t_{i,j,k-1}, \theta) - S_L(t_{ijk}, \theta)\right) \qquad \text{[B.8]}$$

B.4 Confidence intervals

Confidence intervals can be computed on the basis of computing a Hessian matrix and using that to approximate a asymptotic variance-covariance matrix. However, using asymptotic

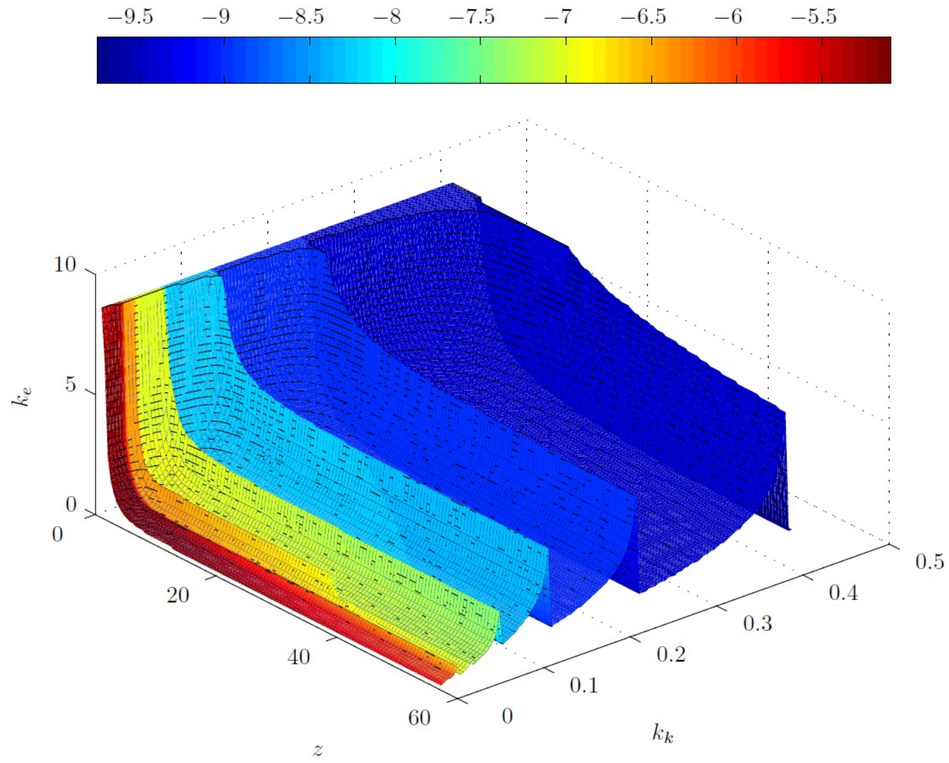


Figure B.3: Maximum loglikelihood for a range of three parameters. For each combination of z and k_k the value for k_e which maximizes the likelihood is chosen. The combination of these three parameters gives the figure its shape; the maximum likelihood in each point is indicated by the used color, NOT by the shape. The maximum likelihood values are displayed on a log-scale: higher numbers (in this case: less negative numbers) indicate higher likelihoods.

theory on small datasets can give incorrect estimates. This may be improved for example by using a Kolmogorov-Smirnov or χ^2 test for goodness-of-fit, or for example by using the Bayesian Information Criterion (BIC) which can warn against overfitting when using many variables in order to achieve a better fit.

B.5 Application

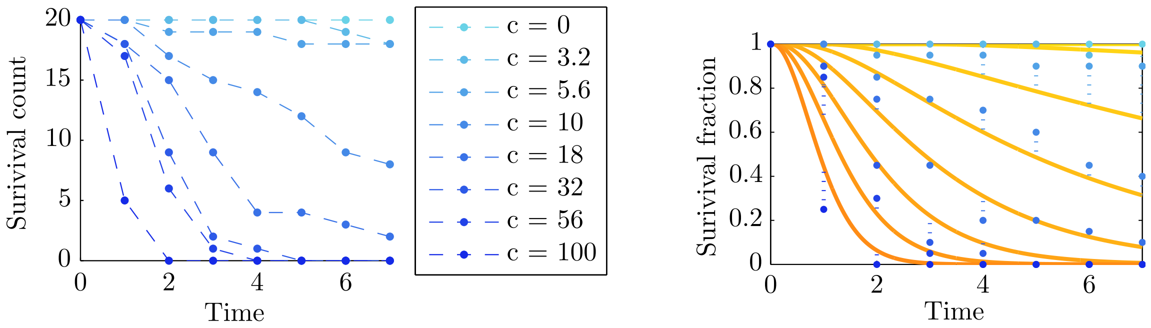
For a simple application, data from Kooijman and Bedaux (1996) was used and fitted to the GUTS model in Table B.1; for this example a SD model was fitted. The fitting process and consecutive use of the fitted model to a fictitious in time-varying concentration profile is shown in respectively Figure B.4a, Figure B.4b and Figure B.4c, Figure B.4d. Maximum likelihood estimates along with standard deviations (based on asymptotic variance-covariance) are given in Table B.2. The model was made in MATLAB; multiple, fast annealing attempts were used to circumvent to try and find the global maximum of the maximum likelihood. The most promising solution from annealing was then further optimised using a simplex algorithm.

	Concentration dieldrin (pesticide) $(\mu g\ l^{-1})$							
Time (d)	0	3.2	5.6	10	18	32	56	100
0	20	20	20	20	20	20	20	20
1	20	20	20	20	18	18	17	5
2	20	20	19	17	15	9	6	0
3	20	20	19	15	9	2	1	0
4	20	20	19	14	4	1	0	0
5	20	20	18	12	4	0	0	0
6	20	19	18	9	3	0	0	0
7	20	18	18	8	2	0	0	0

Table B.1: Survival data of guppies by Kooijman and Bedaux (1996). In the top horizontal row constant concentrations are given, in the first vertical row time is indicated. In the main cells, survival counts are given.

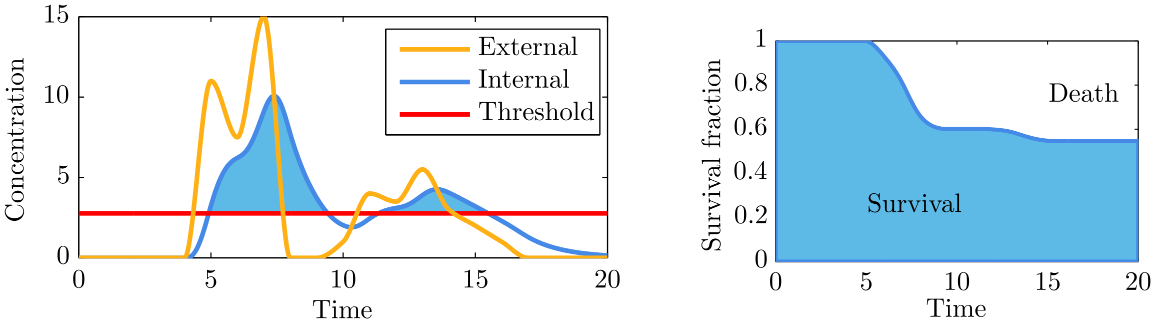
parameter	unit	ML estimate	standard deviation
z	$\mu g\ l^{-1}$	5.20	0.442
k_k	$l\ \mu g^{-1}\ d^{-1}$	0.0376	0.00690
k_e	d^{-1}	0.791	0.195
λ	d^{-1}	0.00835	0.00475

Table B.2: Parameter Maximum Likelihood estimates and standard deviation (the latter based on asymptotic theory)).



(a) *Data collection:* experimental data of a specie exposed to various concentrations.

(b) *Calibration:* finding a common set of model parameters which have the best fit with the experimental data. If enough data is present, perform *validation*.



(c) *Application:* the calibrated model translates an external concentration to a (scaled) internal concentration. If this internal concentration exceeds the threshold (which is one of the calibrated parameters), death is expected.

(d) *Visualization:* estimate of the surviving fraction due to exposure to the variable concentration.

Figure B.4: Typical steps in calibrating (first two images) and applying (last two images) a GUTS model.

Appendix C

Coral Productivity Model

In her master thesis, Larsen (2012) described a coral productivity model which is used in this master thesis. Important for this description are the equations, assumptions and interpretation of these which are described in Section C.5. These are further reviewed, from a probabilistic view, in Section C.6 and Section C.7. In the sections previous to the aforementioned, an overview is given of coral biology and processes. These early sections are a summary of what has been described in Larsen (2012), with some minor additions. The purpose of these sections is solely to give context and background information for the actual model description in Section C.5, because the model tries to schematize a number of these biological processes. ***Please note:*** *Section C.1 up to (and including) Section C.4 should be considered as direct citations of Larsen (2012). Chapter 2, 3 and 4 of Larsen (2012) are the primary sources. A conscious choice was made to include such a lengthy citation in this report, because it describes various parts of coral biology very well, and serves as a necessary introduction into the coral productivity model. Some small additions were made to the citations, but the vast majority remains a direct citation of Larsen (2012). For brevity and readability, quotation marks have been left out.*

C.1 Singapore and corals

The master thesis of Larsen (2012) was written for the faculty Aquatic Science and Technology at the Technical University of Denmark, in collaboration with DHI (Dansk Hydraulisk Institut) Denmark and DHI Singapore. Singapore was found to be a relevant case study due to the presence of coral reefs and naturally presence of relative high background TSS levels. Because the biological nature of the model, application incorporates some biological traits of species specific to Singapore.

C.1.1 General description corals

The tropical reef-building coral (Scleractinia) belongs to the class *Anthozia* of the phylum *Cnidaria*. Jellyfish and sea anemones also belong to *Anthozia*; common for this class is the construction of two skin layers, usually called the ectodermis (external skin layer) and the endodermis (internal skin layer). Corals are sessile (attached; cannot freely move) and therefore have a side which is attached to the substrate (aboral side) and a side exposed to the water (oral side). Skeleton formation occurs in the aboral side, while the oral side contains zooxanthellae (oral ectodermis) and cnidocytes (venomous cells). The venomous cells allow the coral to extend tentacles for defense or capturing prey; the zooxanthellae are able to photosynthesize and allocate energy towards the coral (Larsen, 2012).

C.1.2 Coral morphology

Most corals polyps are colonial, meaning that the coral polyps grow in colonies; these can take many forms. Even within species variations can occur; a morphological variation of a species is called an ecomorph (Veron, 1995). This variation can have a genetic and/or an environmental cause. Genetic variations occurs when different colonies of the same species may have different genotypes, while environmental variation can occur because of morphological plasticity (e.g. physical appearance change due to selective breeding). Due to relatively long lifespans of corals and sporadic spawning events, evolutionary changes in corals are slow (Lasker and Coffroth, 1999; Larsen, 2012). Various lifespans are applicable for a coral reef: reefs themselves are generally thousands of years old; coral colonies, which are part of a reef, can live up to hundreds of years and coral polyps, part of a colony, generally live a few years.

Environmental factors

The most influential environmental factors on intra-specific coral morphology are considered to be light and water movement (Ow and Todd, 2010; Larsen, 2012). Suspended materials, sedimentation rates, temperature, latitudinal effects and nutrient availability also have an effect (Kaandorp et al., 2011; Veron, 1995), but are considered secondary (Larsen, 2012). The rate of calcification in corals is much higher under high light conditions compared with low light conditions. In relatively low light intensities, corals tend to flatten in order to expose as much surface as possible to irradiance (Anthony et al., 2005; Stambler and Dubinsky, 2005; Ow and Todd, 2010; Larsen, 2012). High flow conditions, either due to strong wave action or strong currents, result in slower growth rates and more dense/stronger skeletons (Brown et al., 1985; Scoffin et al., 1992; Larsen, 2012). An example of the difference between high and low flow conditions can be seen in Figure C.1.

Growth forms

Growth forms vary within genus (genus covers a group of species) but, as mentioned above, growth forms also vary within species; variation is even possible within the same colony. The amount of tissue exposed to radiation is very dependent on growth reasons, which is why in the thesis of Larsen (2012) this was regarded as one of the most important traits of coral. Actual division in morphology, or growth forms, is done following the template set by Larsen (2012) and DHI Singapore; other divisions are possible but the most convenient choice is to follow this one. Actual division in morphology, or growth forms, is done following the template set by Larsen (2012) and DHI Singapore; other divisions are possible but the most convenient choice is to follow this one. The list of growth forms can be found below, and pictures showing examples of these growth forms can be found in Figure C.2.

CB Branching corals.

CE Encrusting corals, covering surfaces like rocks, boulders or other corals.

CF Foliose corals, generally more flattened and have a leaf/plate like shape.

CM Massive corals come in many forms: boulders, hemispherical or flattened

CMR Mushroom corals, shaped as a mushroom or a pillow-shaped form. These corals are solitary, but are a significant presence in reefs and therefore mentioned here.

CS Sub-massive corals, a common trait for these corals is the extension of tentacles during both day and night.

These growth forms are grouped together in three groups for calculation purposes later. Group A contains corals which extend their tentacles only during night time. This is a very large and diverse group, but submassive and mushroom corals were considered to be so different from the rest in Larsen (2012) that they got their own groups.

Group A CM, CE, CB, CF

Group B CS

Group C CMR

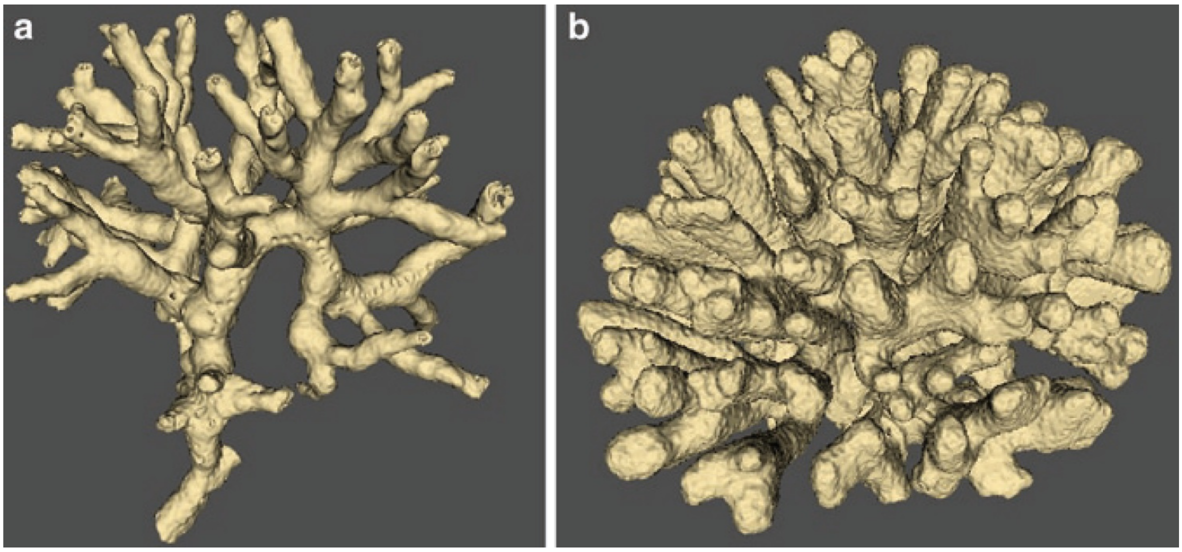
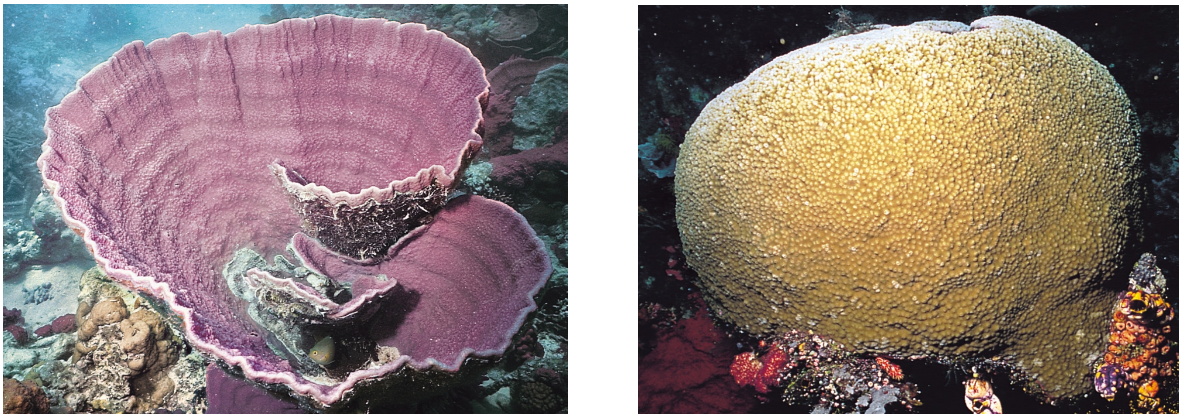
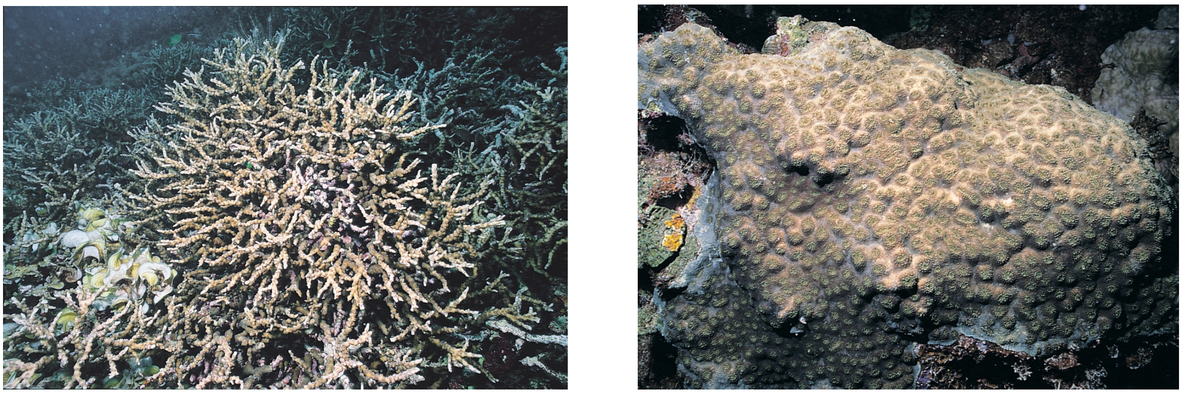


Figure C.1: Surface renderings of CT scans of *Madracis mirabilis*: (a) lateral view of a thin-branching low-flow morph, (b) top view of a compact high-flow morph. Both colonies are visualized on the same scale. The dimensions of object a and b are, respectively, 13 × 8 × 11 cm and 11 × 10 × 6 cm. Description and image taken from Kaandorp et al. (2011).



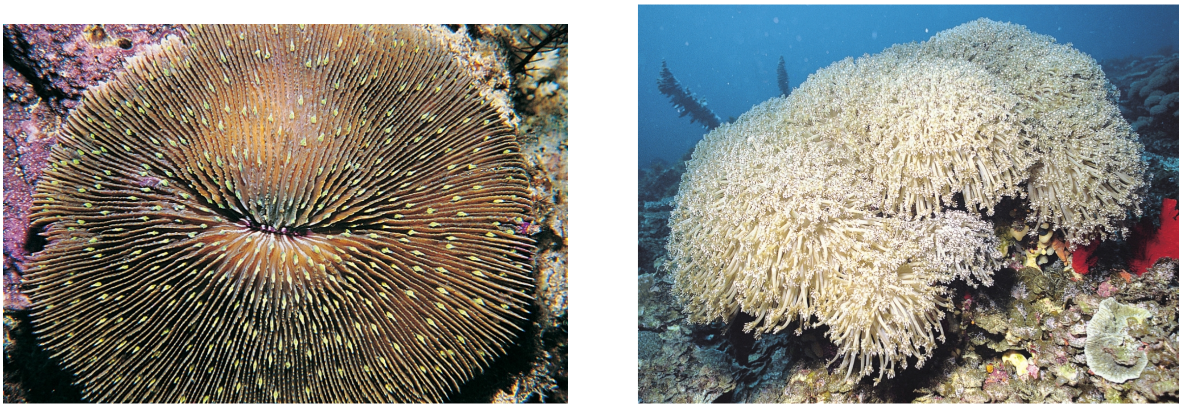
(a) Foliose coral (CF): *Montipora capriconis*.

(b) Massive coral (CM): *Astreopora myriophtalma*.



(c) Branching coral (CB): *Echinopora horrida*.

(d) Encrusting coral (CE): *Acanthastrea rotundoflora*.



(e) Mushroom coral (CMR): *Fungia fungites*.

(f) Submassive coral (CS): *Goniopora columna*.

Figure C.2: Examples of different coral growth forms, taken from Veron and Stafford-Smith (2012); Larsen (2012).

C.2 Ecology of coral reefs

A coral reef is characterized by the presence of large reef-building stony corals (hermatypic corals); these corals grow in colonies of varying shape and structure (Larsen, 2012). Reefs can be divided into fringing reefs, barrier reefs, patch reefs and atoll reefs, as originally proposed by Darwin in 1842 but still in use today (Darwin, 1976; Kaiser et al., 2011; Spalding et al., 2001; Larsen, 2012). Fringing reefs are formed form the coastline and consist of a shallow platform extending seawards ending with a sharp edge sloping in deep water. Barrier reefs are found further from the coast and are generally older. A lagoon separates the barrier reef from the shore. They can develop off shore, but are more often old fringing reefs disconnected from the shore due to sea level rise. Patch reefs are small reefs usually found in lagoons or other shallow locations. Atolls are board circular reefs with a lagoon in the center: these are oceanic reefs origination from weathered volcanic islands (Larsen, 2012).

Despite being called coral reefs, corals are rarely the most abundant actor in a coral reef; but they are the key stone species. Macro algae are very abundant and important for their capability to stabilize the reef structure and serve as a source of food for many other inhabitants. Examples of other inhabitants include, but are not limited to, soft corals, sea anemones, molluscs: the applicable range goes from very small (micro fauna) to very large, including fish (Dubinsky and Stambler, 2010; Larsen, 2012).

C.2.1 Disturbances

Disturbances come from widely varying sources. The most obvious disturbances are not treated here, such as direct damage by trawling fish nets. The disturbances mentioned below, in the context of this thesis, could be defined as generally negative *exposures*, with the exception of coral bleaching which is an *effect*. See also Section 2.4 for a more formal depiction.

Competition

Coral primarily compete in three ways (Chadwick and Morrow, 2011): biologically (fast reproduction to optimize substrate utilization (Chadwick-Furman et al., 2000)), physically (skeleton overgrowth, move growth forms away from competitors, intercept food which would otherwise reach competitors (Chadwick and Morrow, 2011)) and chemically (venomous cells) (Larsen, 2012). See also Figure C.3a.

Phase shifts

Macro-algae are abundantly present on coral reefs. They compete with corals for, among other things, substrate, nutrients and light (Odum and Odum, 1955). Should coral cover decrease, for some reason, it has been observed that a phase shift from coral dominated to algae dominated can occur (Dubinsky and Stambler, 2010; Larsen, 2012). Once in this new state, bringing conditions back to what they where when the reef was coral dominated might not be enough to actually return to a coral dominated state: considerable better conditions could be required to move the system to the old state. Figure C.3b attempts to depict competition between algae and coral.

(a) Corals competing for the same substrate (Chadwick and Morrow, 2011).

(b) Macro-algae growing over coral (Larsen, 2012).

Figure C.3: Examples of competition and a (possible?) phase shift (Larsen, 2012).

Disease and predation

Corals are subjected to diseases such as white or black band disease, bacterial infections, parasites and tumors (Rosenberg and Kushmaro, 2011; Larsen, 2012). Evidence suggest these events are significantly more present in stressed corals than healthy corals (Richmond, 1993; Larsen, 2012). The Crown-of-Thorns starfish *Acanthaster plancii* graze on corals. Under normal conditions, this does not have to be a significant problem, but if conditions change which favor peaks in Crown-of-Thorns starfish density, they can destroy large areas of coral reef. Favorable conditions are currently considered to be nutrient enrichment and/or removal of their main predators due to fishing (Kaiser et al., 2011; Richmond, 1993; Larsen, 2012). See Figure C.4a for a picture of Crown-of-Thorns starfish.

Temperature

Global temperatures are not constant through the years; trends in recent history indicate temperature is currently rising. The why and how much are still debated and not treated here; about one of the few things most parties can agree upon is that temperature is not constant and probably will (continue?) to rise in the future. Any temperature difference also has an effect on the temperature of the world's oceans (Larsen, 2012). Structural temperature changes can change entire ecosystems, while, more specifically for corals, periodic or incidental temperature changes are often associated with bleaching:see also Section C.2.1.

Ocean acidification

Changes in elevated atmospheric CO_2 affect chemical reactions in the ocean. Elevated atmospheric CO_2 levels will lead to elevated CO_2 levels because oceans at least partly absorb the atmospheric levels. CO_2 will react with water (H_2O) to form bicarbonate (HCO_3^-) and hydrogen ions (H^+). The required bicarbonate can be formed by dissolving the calcium carbonate skeleton of corals (Scott C. Doney et al., 2009; Erez et al., 2011; Larsen, 2012). The equations are shown in [C.1].

$$CO_{2;atmos} \rightleftharpoons CO_{2;aq} + H_2O \rightleftharpoons H_2CO_3 \rightleftharpoons H^+ + HCO_3^- \rightleftharpoons 2H^+ + CO_3^{2-} \qquad \text{[C.1]}$$

Even though this seems rather straightforward, actual rate of dissolving calcium carbonate depends on the concentration of calcium and carbonate ions, temperature, salinity, pressure and the particular mineral phase (Scott C. Doney et al., 2009). Furthermore, there are reports indicating that corals can at least partly remedy a drop in pH level (McCulloch et al., 2012).

Coral bleaching

Corals get their pigmentation from their zooxanthellae: if these are expelled the coral appears to be white, which is called coral bleaching. Thermal stress is primarily considered as the main cause of bleaching, but other factors such as salinity, sedimentation and UV radiation could also cause bleaching (Dubinsky and Stambler, 2010; Larsen, 2012). These environmental stresses damage the photosynthetic process resulting in the overproduction of oxygen radicals. This can eventually lead to cellular damage in the zooxanthellae and/or the coral, and can lead to expelling zooxanthellae; effectively breaking the symbioses at least temporarily (Baker et al., 2008). For an example picture of bleaching, see Figure C.4.

Seasonal variation in water temperature is, in most cases, assumed to be not a problem for corals. If the water temperature rises above the long-term maximum for an extended period, coral bleaching is probable. The most severe bleaching events are typically accompanied with coupled ocean-atmosphere phenomena such as El Niño (Baker et al., 2008).

Bleaching is increasingly considered to be a mechanism of coral to expel their current batch of zooxanthellae, and replace them with a “better fitting” type; usually this means more heat resistant (Berkelmans and van Oppen, 2006). However, results found so far have been contradicting: some coral species have been found to exhibit this behavior, while others haven’t. Adaptability of both the coral host and zooxanthellae has also been considered; however preliminary results point to zooxanthellae having the greater potential for adaptability relative to their coral hosts (Császár et al., 2010). On a more global scale, temperature has been found to correlate reasonably well with bleaching; see also Subsection 4.3.1.

(a) Crown-of-Thorns starfish grazing on coral.

(b) Bleached massive coral and partially bleached branching coral.

Figure C.4: Crown-of-Thorns starfish grazing and bleached coral (Larsen, 2012). Images retrieved from http://e-atlas.org.au/ on 29 August 2012.

Sedimentation

Sediments, when suspended, reduce photosynthesis. When settled it is even possible to completely block photosynthesis and block feeding channels. Corals can remedy this by excreting mucus sheets, brush it off with their tentacles, ingest sediment or inflate polyps to increase their available surface area for irradiation. At best sedimentation only costs some extra energy allocation, though in more severe cases it could lead to tissue necroses, bleaching or even mortality (Riegl and Branch, 1995; Larsen, 2012).

C.3 Overview of the main processes of a coral

Growth of corals can be schematized by means of an energy budget: input minus the output of energy. Energy input (I) is attained from heterotrophic and autotrophic feeding. Energy output contains metabolism, or respiration (R), mucus excretion (M), carbon sequestration (S, capture of carbon: skeleton growth) and additional outputs such as reproduction and predation (B). This has been summarized in a general equation in [C.2], where energy is expressed in terms of organic carbon (C) (Larsen, 2012).

$$\frac{dC}{dt} = I \cdot C - R \cdot C - M \cdot C - S \cdot C - B \cdot C \qquad [C.2]$$

C.3.1 Input

Energy input is in this equation comprised of both autotrophic and heterotrophic feeding. Autotrophic growth of corals originates from carbon fixation of the zooxanthellae and the subsequent energy transfer to the coral. Corals can do heterotrophic feeding on organic matter and plankton. This diversifies their food sources (Edmunds and Davies, 1986) and allows them to adapt to different environmental conditions (Anthony and Fabricius, 2000); this is believed to be one of the main reasons for success of corals in the nutrient-poor water they are usually found in (Johannes, 1974) (along with high levels of internal nutrient cycling, see Muscatine and Porter (1977)) (Larsen, 2012).

Autotrophy

Under very favorable conditions for photosynthesis up to 95% of the photosynthetically fixed carbon can be trans-located from the zooxanthellae to the coral, but is more commonly around 40% (Muscatine et al., 1984). The trans-location can make up to 100% of the metabolic requirements (Falkowski et al., 1984; Muscatine et al., 1981); however under stress this is not case (Anthony and Fabricius, 2000; Larsen, 2012).

Zooxanthellae are symbionts living inside the oral endoderm of the coral. They are dinoflagelates of the genus *Symbiodinium* (Dubinsky and Stambler, 2010; Larsen, 2012). Six different clades (group of species and its descendants) of zooxanthellae are able to live in symbiosis with corals, with success varying on the pairing and actual conditions. Corals can expel zooxanthellae, but zooxanthellae are also able to leave the coral themselves. The leaving of zooxanthellae is what happens during bleaching events (Larsen, 2012). It has been suggested that coral expels zooxanthellae under stress in order to get different zooxanthellae which provide a more optimal symbiosis under stress (Baker, 2001).

After photosynthesizing, zooxanthellae distribute the fixed nutrients into a “lunch box”, which is passed to the coral. The coral uses the remaining nutrients for its metabolic requirements (Falkowski et al., 1984; Larsen, 2012). It should be noted this order of events is important, because this effectively says that coral does not feed on zooxanthellae (though it might ingest zooxanthellae which have died from natural causes, primarily to free up space); this means coral does not receive autotrophic carbon during shading. This “lunch box” is formally called the HRF (Host Release Factor). This is a compound in the coral tissue which stimulates the release of the nutrients from zooxanthellae to coral (Dubinsky and Stambler, 2010).

During periods with no or very little light, the symbiotic relation can even be reversed: coral can transfer heterotrophically obtained energy and nutrients to zooxanthellae (Anthony and Fabricius, 2000; Larsen, 2012).

Heterotrophy

Heterotrophy, because of the nutrient-poor water corals are usually found in, is considered to be an important input of nutrients such as phosphorous and nitrogen for both coral and zooxanthellae (Muscatine and Porter, 1977; Larsen, 2012). Corals are able to feed on detrital Particulate Organic Matter (POM), Dissolved Organic Matter (DOM) and plankton ranging from picoplankton to mesoplankton (Houlbreque and Ferrier-Pages, 2009; Ferrier-Pages et al., 2011; Larsen, 2012). The feeding is primarily facilitated by producing mucus nets or strands in which small particles are trapped (Anthony and Fabricius, 2000; Larsen, 2012). It has also been reported that corals farm bacteria in mucus as a food source (Huettel et al., 2006; Larsen, 2012). Secondary processes are capturing particles via nematocyst discharges (nematocysts are commonly known as stinging cells carrying venom) and tentacle grabbing (Ferrier-Pages et al., 2011; Larsen, 2012).

There is still much debate about the importance of heterotrophy as energy input. Comparing high light conditions to low light conditions, the latter has a considerable larger share of heterotrophic feeding. For bleached corals, up to 100% of the nutritional uptake can be accounted to heterotrophy (Falkowski et al., 1984; Larsen, 2012). General consensus is that the actual heterotrophic contribution depends on species, autotrophic energy availability and prey abundance (Houlbreque and Ferrier-Pages, 2009; Hoogenboom et al., 2010b; Larsen, 2012).

C.3.2 Output

Output is, as represented by the equation in [C.2], comprised of metabolism, mucus release, carbon sequestration and additional outputs such as reproduction and predation (Larsen, 2012). A few of these will be further expanded below.

Respiration

Respiration is the amount of carbon lost due to metabolic activities. Respiration is assumed to be mainly dependent on temperature, but other factors such as light or prey availability might also have an effect on respiration (Anthony and Connolly, 2004; Larsen, 2012).

Mucus release

Mucus release functions both as a method for heterotrophic feeding, as well as a method to remove sedimentation. Sedimentation can smother and clog up feeding organs if not removed (Richmond, 1993). In an attempt to quantify the influence of sedimentation on mucus release, Riegl and Branch (1995) investigated the relation between mucus and sedimentation by looking at corals under various light levels with and without sedimentation. From the article it seems that no extra turbidity was introduced due to the sedimentation. The light conditions tested are given as percentages of the subsurface PAR irradiation and amounted to 50% and 25% PAR irradiation. The amount of sedimentation applied was approximately $200mg/cm^2$. A quick and dirty conversion to layer thickness would be (assuming the $200mg/cm^2$ is the dry weight, average sediment dry weight is $1600mg/cm^2$ and a flat surface): $\frac{200mg/cm^2}{1600mg/cm^3} \approx 1.3mm$. The results showed that almost all tested species showed:

- Increased mucus release, also leading to a relatively larger share of respiration belonging to mucus release (increasing roughly from 35% to 65%)
- Significant decrease in photosynthetic production (decreases ranging on average between 45% and 53% depending on the light condition and species)
- Respiration also decreased roughly between 10% and 20%
- A shift from autotrophic to heterotrophic feeding behavior. This was explained with the P/R (Production over Respiration); if the ratio is smaller than one, heterotrophic feeding is expected. This occurred during sedimentation with various light levels: without sedimentation ratios remained greater than one.

"Sediment affects coral metabolism by decreasing photosynthetic production, increasing relative respiration and increasing carbon-loss through greater mucus output", Riegl and Branch (1995). This model is generally accepted (Wesseling et al., 2011; Philipp and Fabricius, 2003; Anthony and Connolly, 2004). In further research done by Stafford-Smith (1993), coral that wasn't successful in clearing the sediment showed (partial) mortality, though heavily species dependent, in the order of a few days. Sediment-rejection efficiency and sediment tolerance were not found to be directly related, though it was found that coarser sediment (500 to 1000 μm) was harder to get rid of than fine sediment (63 to 250 μm).

Reproduction

The coral life cycle consists of two phases; a short planula larval phase and the dominant benthic polyp phase. Reproduction occurs during the polyp phase, and can occur as asexual and cycles of sexual reproduction (Dubinsky and Stambler, 2010; Larsen, 2012). Both sexual and asexual reproduction have a number of modes, and each mode can lead to colony growth or a new colony. In the following list, an overview is given. For asexual reproduction, those entries marked with an asterisk (*) lead to colony growth; the other items lead to new colonies (Cairns, 1988; Larsen, 2012).

Both of the asterisk marked modes (intra- and extratentacular budding within a colony, also the most most common modes of asexual reproduction) are the result of the growth of polyps in the colony. Intratentacular is the internal division of existing polyps, extratentacular is the growth of new polyps adjacent to existing polyps (Dubinsky and Stambler, 2010; Larsen, 2012); the new polyps are connected with older polyps via nerves and muscle in the soft tissue (Parrin et al., 2010; Dubinsky and Stambler, 2010; Larsen, 2012).

New colonies are formed by various modes, where fragmentation/regeneration are common and important modes; the relative large size of fragments (compared to the size of larvae) reduces the high mortality rates normally found in larval and juvenile life stages. Fragments contain both soft tissue and skeleton. Fragmentations can be caused by storms and waves for example (Highsmith, 1982; Dubinsky and Stambler, 2010; Larsen, 2012).

Fragmentation can be non-accidental as well; this happens mostly in solitary corals and occurs due to genetic programming (Cairns, 1988; Larsen, 2012). Another form of fragmentation is due to partial mortality for some reason followed by the creation of new separate colonies (Highsmith, 1982; Larsen, 2012). This can happen both intra- and extratentacular, though in the case of extratentacular polyp bail-out may occur as well: individual polyps detaching due to unfavorable conditions (Sammarco, 1982; Larsen, 2012).

The slanted modes in asexual reproduction are much less prominent modes and are just mentioned for completeness; for more information regarding these modes see for example Cairns (1988).

Asexual Reproduction

- I Intratentacular budding
  - A Budding within a colony*
  - B Fragmentation/regeneration
    - 1 Accidental/traumatic fragmentation
    - 2 Non-accidental fragmentation
    - 3 Partial colony mortality
- II Extratentacular budding
  - A Budding within a colony*
  - B Fragmentation/regeneration
    - 1 Accidental/traumatic fragmentation
    - 2 Partial colony mortality
  - C *Polyp balls*
  - D *Anthoblasts*
- III *Transverse division*
- IV *Asexual planulae*

Sexual Reproduction

- I Hermaphroditic
  - A broadcast spawner
  - B brooder
  - C mixed pattern
- II Gonochoric
  - A broadcast spawner
  - B brooder
  - C mixed pattern

Sexual reproduction also has several modes: roughly in order of most common to least common they are hermaphroditic broadcast spawning, gonochoric spawning, hermaphroditic brooding, gonochoric brooding, hermaphroditic mixed patterns and gonochoric mixed patterns. The most common modes require external fertilization, and thus spawning synchronicity is considered to be of great importance (Guest et al., 2002, 2005b; Dubinsky and Stambler, 2010; Larsen, 2012). Temperature variance seems to be one of the key triggers, even for equatorial reefs (Guest et al., 2005a; Larsen, 2012). Factors believed to (predominantly negatively) influence reproductive success include temperature, ultra violet radiation, salinity, turbidity and sedimentation (Kojis and Quinn, 1984; Humphrey et al., 2008; Torres et al., 2008; Randall and Szmant, 2009; Larsen, 2012). Rough estimates approximate that less than 1% of the daily gross photosynthetic budget is used for sexual reproduction (Edmunds and Davies, 1986; Larsen, 2012).

C.4 Existing models

A number of models were reviewed in Larsen (2012). Only some of these models are mentioned here; a full list of reviewed models can be found in Larsen (2012). In Larsen (2012) models were categorized as a balance between realism, complexity and generality. Highly realistic and complex models are usually focused on a single or a few processes and model these well, but often do not take into account other possible influencing factors. An example of such a model is the model by Kaandorp et al. (2003) which models the flow around coral forms in order to deduce an optimal flow velocity belonging to that certain form and can be used to predict the state of the modeled coral colony.

Anthony and Connolly (2004) used a model to examine examine physiological niche boundaries. It is posed that any resource can become inhibiting, for example light causing bleaching, higher turbidity causing higher light attenuation. They tried to find the physiological niches, or range of favorable conditions, in which certain coral species will thrive (Larsen, 2012). The model uses energy from irradiation and heterotrophy as resources, while respiration and organic carbon losses (mucus production) are modeled as losses. These four variables are all initially modeled as being dependent on both turbidity and light, except for heterotrophy which is modeled to be dependent only on turbidity. This means irradiation and turbidity both influence resources and losses. Uncommon among similar models is the variation in maintenance costs for different resource regimes; most models only vary maintenance costs dependent on organism age and size. It should be noted that the model seemed fairly insensitive to changes in heterotrophic, which seems to contrast earlier found results of the significance of heterotrophy (among others Falkowski et al. (1984); Anthony and Fabricius (2000)).

COMBO, or Coral Mortality and Bleaching Output, is a relatively simple, general model which works on both temporal and spatial large scales (decades to centuries and spatially) and was created to predict coral mortality from temperature and CO_2 stress. It proved to be capable of predicting the mass coral bleaching event in 2005 in the Caribbean. However, in another application it was noted that the physiological adaptability of the corals (selection of symbionts, selection of more resilient strains) has a significant influence on the long term predictions; and unfortunately this factor is largely unknown (Buddemeier et al., 2008, 2011; Hoeke et al., 2011; Larsen, 2012).

CORSET, or Coral Reef Scenario Evaluation Tool, is another large scale model, put forward as a regional management support tool. It models spawning and brooding corals, macroturf, macroalgae, fish (herbivorous and piscivorous), and sea urchins. It models growth, competition and recruitment, and the model is forced with fishing-pressure, destructive fishing activity, damaging storms, coral bleaching disease, crown-of-thorns starfish outbreaks, nutrification and sedimentation. CORSET predicts the relative consequences of different management scenarios on a very large scale (100s of kilometers), and is a very general model (Melbourne-Thomas et al., 2011a,b; Larsen, 2012).

C.5 Productivity model: Schematization and equations

In her thesis, Larsen (2012) proposed a coral productivity model forced by turbidity for management purposes specific for use in Singapore. The output of the model is primary in terms of carbon content, but an attempt was made to translate this in coral cover; this is because coral surveys generally measure coral cover. The general energy budget was already mentioned in equation [C.2], but is repeated here and graphically shown in Figure C.5:

$$\frac{dC}{dt} = I \cdot C - R \cdot C - M \cdot C - S \cdot C - B \cdot C$$

On a high level, the model as proposed by Larsen (2012) only models the carbon production of coral due to irradiation; light is seen as the primary proxy of coral well-being. A global overview of the steps taken in the model is given in Figure C.6.

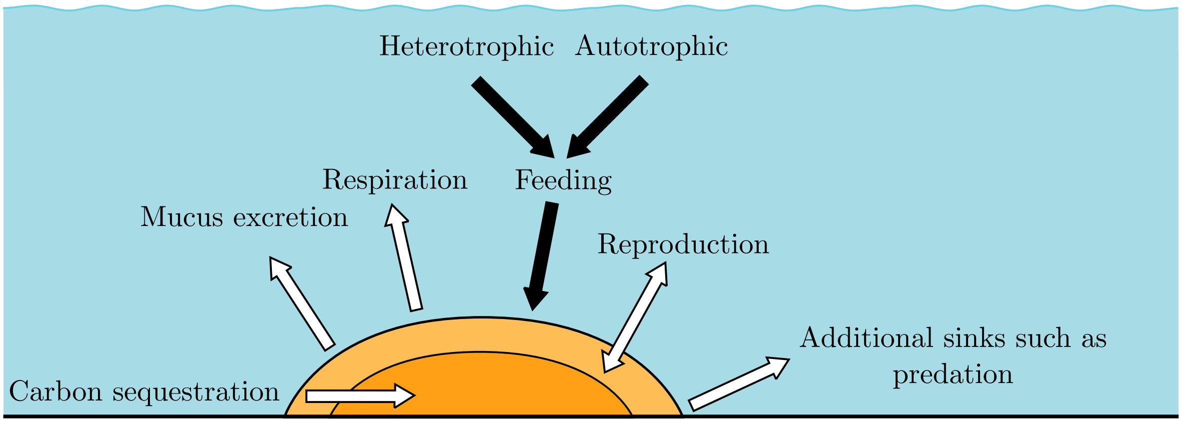


Figure C.5: Graphical representation of equation [C.2]. The light arrows indicate energy output, the dark arrows indicate energy input. The light arrows that are pointed inwards indicate an eventual contribution to growth of the coral.

C.5.1 Production and irradiance

As shown in Figure C.6, the model starts with a subsurface irradiation (I_0), which is transformed to a local irradiation ($I(z)$) at the depth of the coral reef. This depth is the distance between the subsurface and the location of the coral reef: this means the depth (z) is determined largely by tides. The reduction of light is further influenced by the attenuation of matter in the water: this could be for example algae and sediments. Larsen (2012) used equation [C.3], where TSS ($K_{TSS} \cdot TSS$) and some background constant (K_B) are, together with depth, assumed to influence the local irradiance:

$$I(z) = I_0 \cdot e^{-(K_B + K_{TSS} \cdot TSS) \cdot z} \qquad \text{[C.3]}$$

A more detailed description of irradiance and this relation can be found in Appendix D. With the local irradiance available, production of carbon or oxygen can be determined by means of production-irradiance curves (P/I curves) as shown in Figure C.7.

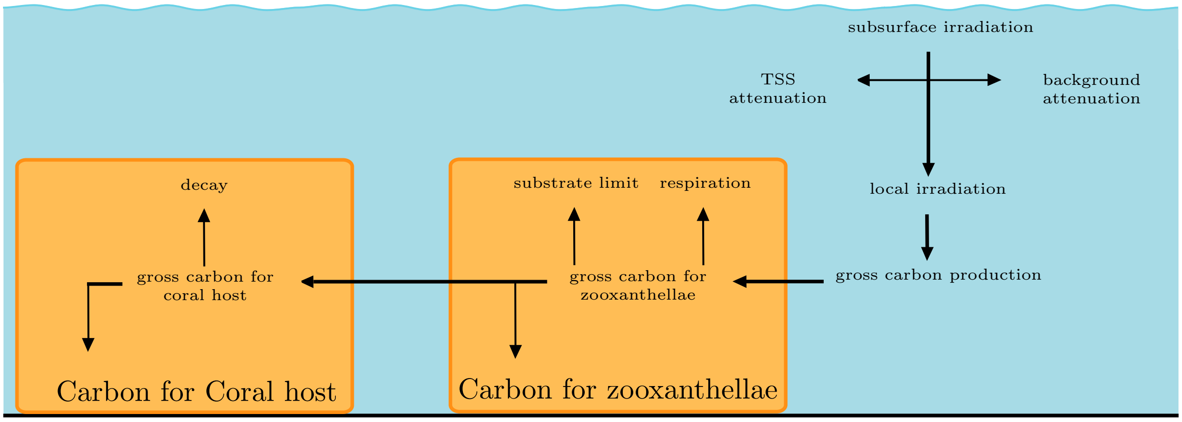


Figure C.6: Graphical representation of the calculation scheme followed in the coral productivity model by Larsen (2012).

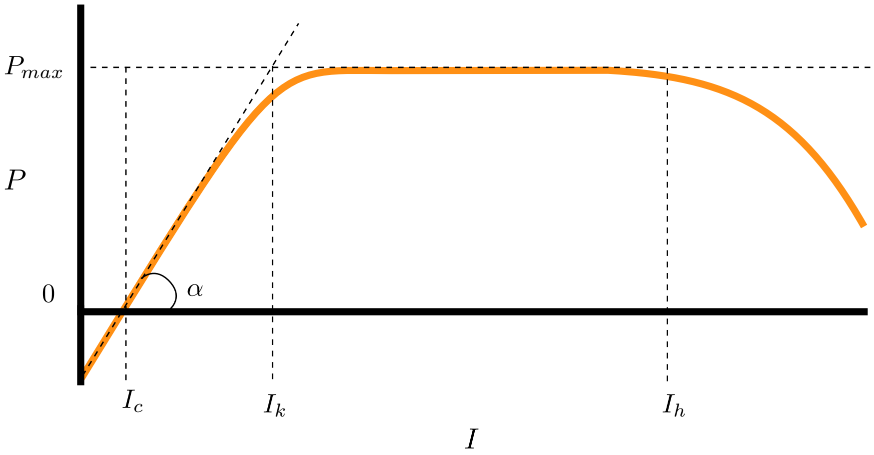


Figure C.7: Production-irradiance curve showing gross primary production (P) as a function of irradiance (I). I_c is the compensation point where production equals respiratory needs, I_k is the saturation point, I_h is the photo-inhibition point, P_{max} is the maximum level of photosynthesis and α is the photo efficiency. Description and image after Larsen (2012).

As shown in Figure C.7, there is a positive correlation between irradiance and production, until maximum production is reached. If irradiation continues to increase beyond this point, photo-inhibition can be reached where production starts to decline again because the irradiation has reached an intensity where it becomes detrimental to the production. See also Section C.2.1. Because of the turbid conditions in Singapore, photo-inhibition was considered not to be an issue in Larsen (2012). However, this says nothing about the likelihood of attaining photo-saturation: this can be expected to occur.

P/I Curves can come in a number of expressions, but Jassby and Platt (1976) found that the equation in [C.4] fitted experimental data best, which was verified for corals by Chalker (1981):

$$P(I)_{gross} = P_{max} \cdot tanh\left(\frac{\alpha \cdot I(z)}{P_{max}}\right) \tag{C.4}$$

This is actually an alternative formulation of the one found in Chalker (1981), the original looked like the equation in [C.5]:

$$P(I)_{gross} = P_{max} \cdot tanh\left(\frac{I(z)}{I_k}\right) \tag{C.5}$$

Equation [C.4] is easier to use in calculations, but contains a commonly used approximation where the angle α (in radians) is equaled to P_{max}/I_k as opposed to $tan\,(P_{max}/I_k)$; this approximation is actually quite good as long as α is (much) smaller than one. Both equations result in gross production estimates; taking into account respiratory requires a minor modification of [C.4] into the equation shown in [C.6] (Schutter, 2010):

$$P(I)_{nett} = R_{dark} + P_{max} \cdot tanh\left(\frac{\alpha \cdot I(z)}{P_{max}}\right) \tag{C.6}$$

C.5.2 Zooxanthellae production

Growth of coral is, aside of available light, dependent of the available substrate. Corals are benthic (living on the bottom), which lead Larsen (2012) to roughly estimate 1 m^2 of suitable substrate can accommodate 1 m^2 of coral; or 1 m^2 of zooxanthellae. This is one of the fundamental assumptions of the production model by Larsen (2012). This is implemented using a Michaelis-Menten half-saturation expression as shown in [C.7]:

$$\text{density limitation} = ZX_{max}\frac{ZX}{ZX + k_{ZX}} \tag{C.7}$$

In this equation, k_{ZX} is a measure of the inverse of the efficiency of the production. For the case of Singapore, it is assumed that nutrient availability and temperature are not limiting the photosynthetic production; this assumption was made because of the relative stable temperature regime in Singapore and high levels of internal nutrient recycling. Other factors such as salinity and pH might also affect production, as these individually or combined with temperature might cause bleaching, but are also left out in this model (Larsen, 2012). Combining [C.4] and [C.7] leads to the photosynthetic production equation in [C.8] for zooxanthellae.

$$phot = P_{max} \cdot tanh\left(\frac{\alpha \cdot I(z)}{P_{max}}\right) \cdot ZX_{max}\frac{ZX}{ZX + k_{ZX}} \tag{C.8}$$

C.5.3 Production allocation

Equation [C.8] describes the gross amount of photosynthetic product available for the zooxanthellae to respire and pass along to the coral host. Larsen (2012) proposed to describe the division between coral and zooxanthellae as [C.9]:

$$
\begin{aligned}
phot &= ZX_{phot} + CC_{phot} \\
ZX_{phot} &= \begin{cases} (1 - maxAllo) \cdot phot & \text{if } \frac{ZX}{CC} > maxDens \\ (1 - allo) \cdot phot & \text{Otherwise} \end{cases} \\
CC_{phot} &= \begin{cases} maxAllo \cdot phot & \text{if } \frac{ZX}{CC} > maxDens \\ allo \cdot phot & \text{Otherwise} \end{cases}
\end{aligned}
\tag{C.9}
$$

Net carbon products for coral and zooxanthellae are the gross products mentioned in [C.9], minus any carbon requirements by either coral or zooxanthellae. This was defined by Larsen (2012) as [C.10]:

$$
\begin{aligned}
CC &= CC_{phot} - CC_{decay} \\
ZX &= ZX_{phot} - ZX_{resp} \\
CC_{decay} &= CC_{mort} \cdot CC_{t;aff}^{\left(T - CC_{t,ref}\right)} \cdot CC \\
ZX_{resp} &= ZX_{darkR} \cdot ZX_{Q10}^{\left(\left(T - ZX_{t,ref}\right)/10\right)} \cdot ZX
\end{aligned}
\tag{C.10}
$$

The difference between nett and gross carbon zooxanthellae products are here assumed to be caused by dark respiration, modified by a metabolic temperature dependency (ZX_{Q10}), which itself is modified by the current temperature relative to a set reference temperature. For coral the equation looks the same, but instead of respiration, mortality is taken which is modified by the affinity of coral to die due to temperature differences from a set reference temperature.

C.6 Probabilistic calculation

C.6.1 Stochastic variables estimates

Production and irradiance

For production and irradiance, the following equations are proposed:

$$
\begin{aligned}
I(z) &= I_0 \cdot e^{-(K_B + K_{TSS} \cdot TSS) \cdot z} \\
P(I)_{gross} &= P_{max} \cdot tanh\left(\frac{\alpha \cdot I(z)}{P_{max}}\right)
\end{aligned}
$$

In this equation, I_0 (subsurface irradiation) is modeled as a stochastic variable; the standard deviation should represent the natural occurring variance in the daily light availability. No usable information was present to instead use a time-varying signal, which resulted in the representation of the light availability as a stochastic variable. It is acknowledged that this is a serious simplification, because light probably varies seasonally, and is tied to weather conditions.

TSS is comprised of two components: a background TSS value, modeled as a stochastic variable, and a dredging induced component which is a time-varying signal. The depth z is the height of the water column above the point of interest; this component is also divided in two components. The first component is a random variable indicating the uncertainty of the depth profile of a coral reef. The second component is a tidal water-level time-varying signal. Both dredging induced TSS and tidal water-level z usually come from output of programs such as MIKE 21/3 or Delft3D.

- I_0 is estimated to be $25E/\left(m^2 \cdot d\right)$ (Larsen, 2012). Standard deviation has been guessed to be in the order of 5, in an attempt to capture the expected great variability in daily irradiance.
- K_B is, as in Larsen (2012), estimated to be 0.2649. The standard deviation is roughly estimated to 0.05. See also Appendix D.
- K_{TSS} is, as in Larsen (2012), estimated to be 0.063. The standard deviation is roughly estimated to 0.005. See also Appendix D.
- Background TSS is, as in Larsen (2012), estimated to be 6 mg/L. The standard deviation is roughly estimated to 1. See also Appendix D.
- The depth varying point of interest part of z is estimated to have a mean of 0 and a standard deviation of 0.25. See also Appendix D for a motivation.

The second formula estimates production rates, which is discussed in more detail in Appendix D. In Larsen (2012), estimates are made for all three the groups mentioned in Section C.1.2. For this part (showing final uncertainty surrounding an answer) it is sufficient to show only a single group. Group B has been chosen for further calculations, and the parameters are mentioned in Table C.1. In most cases, oxygen parameters are measured but more interesting are carbon related parameters; these are converted from the measured oxygen parameters.

Parameter	**Value**	**Unit**
$P_{max;O2}$	8.83	$\left[\frac{\mu mol O_2}{cm^2 hr}\right]$
$R_{dark;O2}$	2.34	$\left[\frac{\mu mol O_2}{cm^2 hr}\right]$
α_{O2}	0.017	$\left[\frac{\frac{\mu mol O_2}{cm^2 hr}}{\frac{\mu mol foton}{m^2 s}}\right]$
$P_{max;C}$	11.6	$\left[\frac{gC}{cm^2 d}\right]$
$R_{dark;C}$	5.38	$\left[\frac{gC}{cm^2 d}\right]$
α_C	30	$\left[\frac{gC}{E}\right]$

Table C.1: Carbon production parameters as used in Larsen (2012).

The values in Table C.1 are literally from Larsen (2012) and according to this document, the oxygen values were converted to carbon values using equation [C.11] from Muscatine et al. (1981):

$$P_c = P_{O2} \cdot 0.375 \cdot PQ_z^{-1} \qquad \text{[C.11]}$$

In this equation, 0.375 appears to be the molar weight of carbon divided by that of oxygen: $12/(2 \cdot 16)$. PQ_z is the photosynthetic quotient (difference between produced and actually assimilated photosynthesis), and assumed to be 1.1 (Muscatine et al., 1981). However, if this conversion is done, different numbers are obtained: for the conversion, see Table C.2 and for the obtained answers see Table C.3.

Unit conversion	**Factor**	**Notes**
$\left[\frac{\mu molO_2}{cm^2hr}\right]$ to $\left[\frac{gC}{cm^2d}\right]$	$\frac{10^{-6}}{10^{-4}} \cdot 12 \cdot 32 \cdot \frac{0.375}{1.1} \approx 1.31$	12 stands for the number of hours in a day expected to have sunlight, 32 for the molar weight of oxygen.
$\left[\frac{\mu molO_2}{cm^2hr}\right]$ to $\left[\frac{gC}{cm^2d}\right]$	$\frac{10^{-6}}{10^{-4}} \cdot 24 \cdot 32 \cdot \frac{0.375}{1.1} \approx 2.62$	24 stands for the number of hours in a day.
$\left[\frac{\frac{\mu molO_2}{cm^2hr}}{\frac{\mu molfoton}{m^2s}}\right]$ to $\left[\frac{gC}{E}\right]$	$\frac{3600^{-1}}{10^{-4}} \cdot 32 \cdot \frac{0.375}{1.1} \approx 30.3$	E (einstein unit) stands for mole photons regardless of frequency, 3600^{-1} is the conversion from seconds to hours.

Table C.2: Conversion for the three parameters mentioned in Table C.1.

Parameter	**Value**	**Unit**
$P_{max;O2}$	8.83	$\left[\frac{\mu molO_2}{cm^2hr}\right]$
$R_{dark;O2}$	2.34	$\left[\frac{\mu molO_2}{cm^2hr}\right]$
α_{O2}	0.017	$\left[\frac{\frac{\mu molO_2}{cm^2hr}}{\frac{\mu molfoton}{m^2s}}\right]$
$P_{max;C}$	11.6	$\left[\frac{gC}{cm^2d}\right]$
$R_{dark;C}$	6.13	$\left[\frac{gC}{cm^2d}\right]$
α_C	0.52	$\left[\frac{gC}{E}\right]$

Table C.3: Carbon production parameters calculated using the factors in Table C.2.

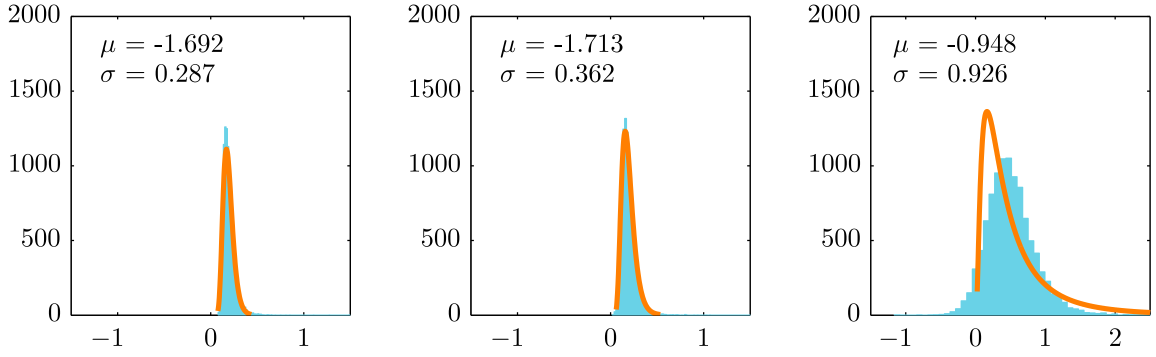
$P_{max;C}$ agrees quite well, but $R_{dark;C}$ shows a slightly higher answer. This could either be a calculation error, or some not-mentioned factor that reduces the dark respiration. The alpha shows quite a difference, but if the factor for time conversion is reduced to 60^{-1}, the answer becomes 31. This seems to be in agreement with the answer in Larsen (2012), which could indicate either that the conversion was done incorrectly, or that the unit was written down incorrect (seconds instead of minutes).

For the probabilistic calculation in this section, it is deemed sufficient to use the values reported by Larsen (2012); the conversion was done primarily to see what the conversion was about. The conversion is assumed to be a “precise” conversion, even though it could be argued it is not; especially for the PQ_z parameter. Assuming the factors to be point values allow linear transformations of the estimated oxygen parameter standard deviations estimated in Section D.4 to be used and transformed to carbon production standard deviations. It is assumed that the deviations for the values in Table C.1 are approximately the same as those found in Section D.4: 1.00, 1.00 and 0.0100 for $P_{max;O2}$, $R_{dark;O2}$ and α_{O2} respectively. Using the rule for linear transformation for random variables, the variance of a linearly transformed standard deviation is the product of the original variance and the squared linear factor. Because the standard deviation is the square root of the variance, the linearly transformed random variable is the product of the factor and the original standard deviation. This means standard deviations are now 1.31, 2.62 and 18.2, for $P_{max;C}$, $R_{dark;C}$ and α_C respectively with the values mentioned in Table C.1 as means.

These are not yet usable values in the coral productivity model by Larsen (2012); the current set of parameters indicate carbon production per square meter and are not a good proxy for biomass. One square meter of subsurface does not have to equal one square meter of biomass surface. Therefore these values are divided by a TOC (Total Organic Carbon, $\left[\frac{gC}{m^2}\right]$) value. For the species parameters in Table C.1, the TOC value found in Larsen (2012) was found to be 64.66 gC/m^2 with a standard deviation of 15.70. Assuming both the TOC and the carbon parameters are normally distributed with aforementioned means and standard deviations, the final, usable parameters are:

$$\begin{aligned} P_{max} &= P_{max;C}/TOC & \left[d^{-1}\right] \\ R_{dark} &= R_{dark;C}/TOC & \left[d^{-1}\right] \\ \alpha &= \alpha_C/TOC & \left[\left(E \cdot m^2\right)^{-1}\right] \end{aligned}$$

If both TOC and the carbon production parameters are normally distributed, the final parameters do not have to be normally distributed as well. The quotient of two standard normal distributions is a Cauchy distribution; other distribution functions have to be derived. because these parameter estimates are first estimates, a more crude approach by means of a quick Monte Carlo simulation ($n = 10000$) and consecutive distribution fitting (using BESTFIT, which fits and ranks some 18 well-known distributions based on K-S tests, A-D tests or chi squared tests). From this it appeared that a lognormal distribution fitted P_{max} and R_{dark} best, while a normal distribution seemed to be a good fit for α. While the lognormal distribution always produces numbers that are greater than zero, the normal distribution for α can produce negative numbers: this is awkward from a physical perspective. To remedy this all negative numbers in the Monte Carlo were made absolute and a lognormal distribution was fitted. This results in a considerable worse fit, but a physically sane distribution. For all three parameters plots of histograms and distribution fits are shown in Figure C.8.



(a) Logrnormal P_{max} (b) Lognormal R_{dark} (c) Lognormal α

Figure C.8: Histograms and distribution fits for P_{max}, R_{dark} and α.

Zooxanthellae production

Substrate limitation and corresponding maximum zooxanthellae was expressed using the following equation:

$$\text{density limitation} = ZX_{max}\frac{ZX}{ZX + k_{ZX}}$$

ZX_{max} some estimates in literature; Larsen (2012) referred to Muscatine et al. (1981). This seems like a parameter which was estimated based on some closely related parameters, therefore a crude estimate of this parameter for group B has been set to a normally distributed parameter with mean 5.6 and standard deviation 1.

k_{ZX} is difficult to estimate: the substrate limitation formulation in Larsen (2012) was chosen because it made biologically sense, but the parameter choice was chosen to mimic the relation $tanh\left(ZX/ZX_{max}\right)$ in Hoogenboom et al. (2010a). Therefore, the mean has been chosen to be 1, similar to Larsen (2012), but the standard deviation has been set to 0.25 to reflect the error in mimicking the original relation.

Production allocation

$$\begin{aligned} phot &= ZX_{phot} + CC_{phot} \\ ZX_{phot} &= \begin{cases} (1 - maxAllo) \cdot phot & \text{if } \frac{ZX}{CC} > maxDens \\ (1 - allo) \cdot phot & \text{Otherwise} \end{cases} \\ CC_{phot} &= \begin{cases} maxAllo \cdot phot & \text{if } \frac{ZX}{CC} > maxDens \\ allo \cdot phot & \text{Otherwise} \end{cases} \end{aligned}$$

$maxAllo$ and $allo$ describe the “ideal” and “regular” allocation part of photosynthetic products from zooxanthellae to coral host. Both are based on Larsen (2012) and sources cited within; $maxAllo$ is supposed to be close to 1. Because numbers greater than 1 are not physically possible, a choice was made for 1 minus a lognormal distribution with a mean around 0.02 and variance 0.0016. When calculating this the relevant μ and σ parameters of the accompanying distribution, μ and σ are respectively -4.72 and 1.61. For $allo$ a normal distribution with mean 0.4 and standard deviation 0.1 was chosen. $maxDens$ is also based on Larsen (2012) and chosen to be normally distributed around 0.12 with standard deviation 0.05.

The remainder of the parameters contain a few temperature-indicating variables. These were all set to a mean of 29.6 degrees Celsius, and a standard deviation of 0.5 (normally distributed). Because temperature affinity was blocked out in Larsen (2012)during simulation runs (for a number of reasons, lack of data being one), the temperature affinity parameter $CC_{t,aff}$ has been set to a deterministic value of 1. Furthermore, coral mortality CC_{mort} was based on Larsen (2012) as well with mean 0.00875 and standard deviation 0.002 (normally distributed).

The last parameter is the zooxanthellae Q10 factor. mention was made in Larsen (2012) that numbers varied significantly between 2 and 20, with most values around 5 to 8. Staying close to the choices made in Larsen (2012), a normal distribution was chosen with mean 6.5 and standard deviation 2.

C.6.2 Overview of stochastic variables

In Table C.4 an overview of all variables can be found. For this calculation no dredging induced excess TSS is taken into account; only background TSS. The water level variations are tidal induced plus a anticipated reef depth uncertainty. A sample of the tidal variations is shown in Figure C.9.

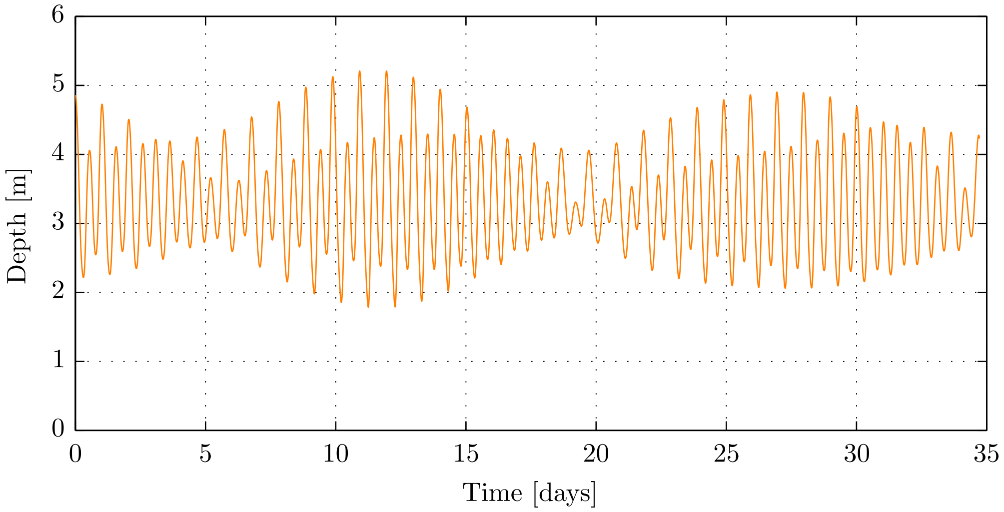


Figure C.9: Tide induced water level variations. Data was extracted from a point near Sultan Shoal (using DelftDashboard) and modified to more closely match the average depth value as found in Larsen (2012), by adding 1.62 meters to the raw data output. This was used as input in the model, believed to be a representable dataset, though still a crude first-order estimate, of water level variations.

Process	**Variable name**	**Symbol**	**Distribution**	**Value**	**Unit**
irradiance	subsurface irradiance	I_0	norm	25, 5	$E/(m^2 \cdot d)$
	background attenuation constant	K_B	norm	0.2649, 0.05	m
	Total Suspended Solids	TSS	norm	6, 1	mg/L
	TSS attenuation constant	K_{TSS}	norm	0.063, 0.005	$1/m$
	depth (reef depth)	z	norm	0, 0.25	$L/(mg \cdot m)$
irradiance, production	local irradiance	I	calc	-	$E/(m^2 \cdot d)$
production	maximum production rate	P_{max}	logn	-1.69, 0.29	$1/day$
	photosynthesis efficiency	α	logn	-0.95, 0.93	$1/(E \cdot m^2)$
	production rate	$P(I)$	calc	-	$1/day$
	(carbon) total photosynthetic products available	$phot$	calc	-	g/m^2
substrate limitation	(carbon) zooxanthellae population density	ZX	determ	1.6*	g/m^2
	(carbon) Coral density	CC	determ	18*	g/m^2
	(carbon) inverse of the efficiency of the production	k_{ZX}	norm	1, 0.25	g/m^2
	(carbon) maximum zooxanthellae population density	ZX_{max}	norm	5.6, 1	g/m^2
	maximum density of ZX / CC	$maxDens$	norm	0.12, 0.05	–
production allocation	maximum allocation to zooxanthellae and coral	$maxAllo$	1-logn	-4.72, 1.61	–
	allocation to zooxanthellae and coral	$allo$	norm	0.4, 0.1	–
CC, ZX carbon loss	temperature	T	norm	29.6, 0.5	$^\circ C$
CC carbon loss	temperature affinity for coral mortality	$CC_{t,aff}$	determ	1	–
	reference temperature background mortality	$CC_{t,ref}$	norm	29.6, 0.5	$^\circ C$
	coral mortality at $CC_{t,ref}$	CC_{mort}	norm	0.00875, 0.002	$1/day$
	(carbon) coral decay	CC_{decay}	calc	-	g/m^2
ZX carbon loss	zooxanthellae Q10 factor	ZX_{Q10}	norm	6.5, 2.5	–
	reference temperature dark respiration zooxanthellae	$ZX_{t,ref}$	norm	25, 0.5	$^\circ C$
	zooxanthellae dark respiration	ZX_{darkR}	logn	-1.71, 0.36	$1/day$
	(carbon) zooxanthellae respiration cost	ZX_{resp}	calc	-	g/m^2

Table C.4: Parameters used in the coral productivity model by Larsen (2012). Parameters marked with an asterisk (*) are only used as initial values and are supposed to have little influence on the end result. Not mentioned are the (both optional) dredging induced TSS and tidal induced varying water levels which are deterministic, but in time varying signals.

C.6.3 Monte Carlo simulation

Using the distributions in Table C.4, a Monte Carlo calculation was set up; results can be found in Figure C.10. From this graph it is apparent that there is a huge amount of variation surrounding the final answer. The 95% confidence interval in Figure C.10b spans from total mortality to biomass numbers four times higher than what would be expected on average.

In short, the variation is much larger than what would reasonably be expected in practice. This makes the model, based on the assumptions and estimates described in the previous sections, not suitable for making absolute statements on changes in coral biomass. Though relative differences, by comparing distributions of biomass, could be a possibility for applying this model.

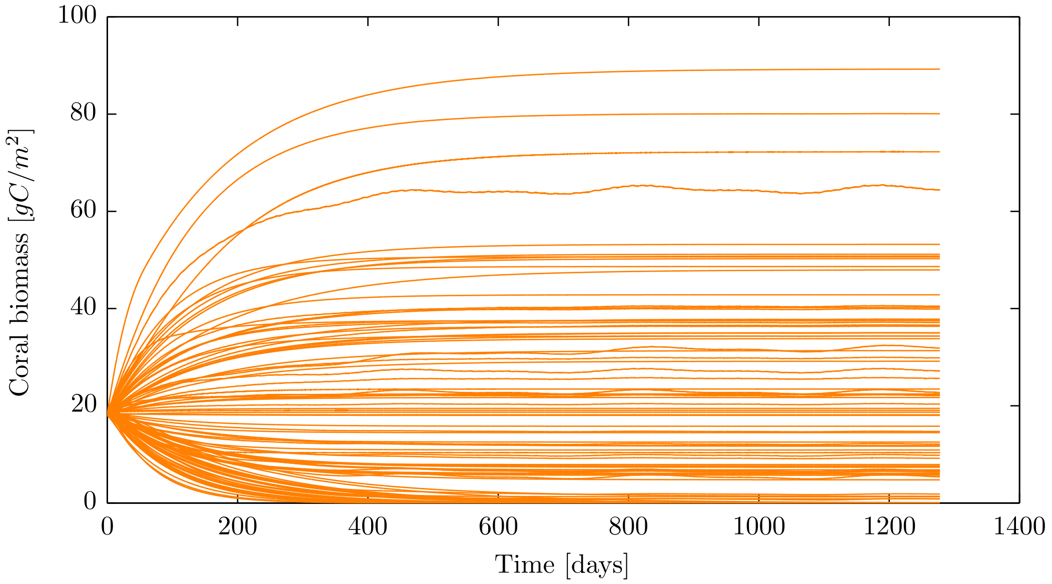
In Table C.5 the alphas of each variable from a FORM (First Order Reliability Method) analysis are shown. These alphas indicate the relative importance of the particular parameter on the final answer: a positive number indicates a positive correlation, a negative number a negative correlation. Even though the productivity model is highly non-linear, reducing the applicability of a FORM method, the alphas are used here as a crude sensitivity analysis. The absolute values can probably not be trusted, though the relative ordering might still have some merit. From the FORM analysis some alpha values are zero. For the initial values of CC and ZX this can be easily explained because these values are only used in the first time step: because of the long time period, the influence of the initial parameters was indeed expected to be zero. Figure C.10b shows that after approximately a year equilibrium biomass values are obtained: if the time period would be set to less than a year the 'spin-up' of the model could start showing its influence. The z parameter is actually only the depth uncertainty of the coral reef depth: why this is zero is a bit more tricky. It could be explained by the $tanh$ shape of the production formula: depth and light combinations exist such that the productivity is always P_{max} because of the asymptotic production formula. This idea is further reinforced by the relative importance of P_{max}. The zero values for the temperature variables were intentionally nullified and were expected to be zero. Finally, the combination of $maxAllo$ and $maxDens$ being zero: the fact that these parameters are both zero is not surprising ($maxDens$ determines if $maxAllo$ or $allo$ is used). Apparently, $maxDens$ is never reached, or only reached during spin-up: the former could be because a density check was already in place by the Michaelis-Menten equation. On the other hand, the importance of P_{max}, ZX_{dark} and ZX_{max} shows that production parameters are very important for this simulation.

I_0	K_B	K_{TSS}	TSS	z	P_{max}	α
2.9E-03	-4.4E-03	-2.5E-03	-5.8E-03	0	6.2E-01	8.6E-03

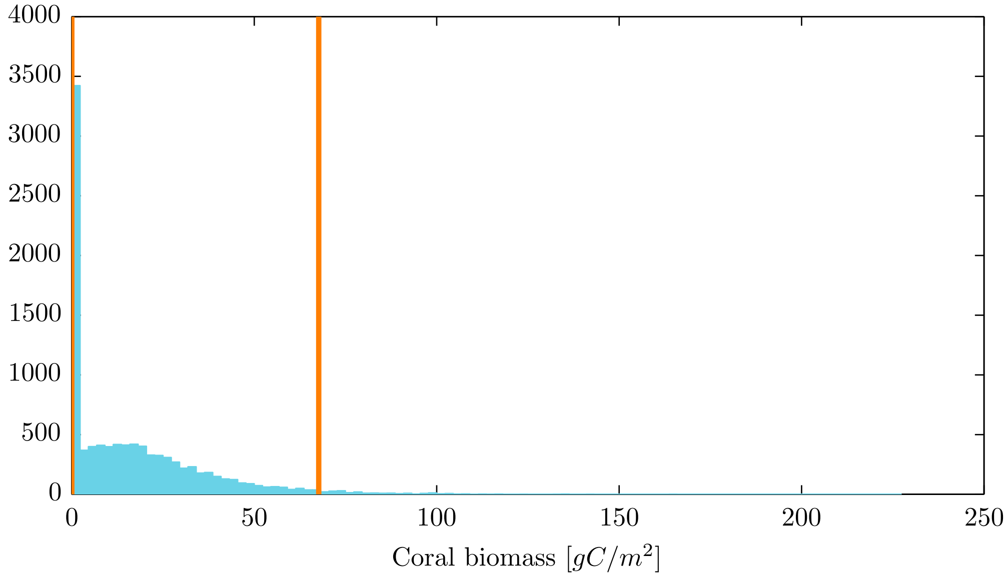
CC_{mort}	$CC_{t,aff}$	T	$CC_{t,ref}$	CC_{init}	ZX_{darkR}	ZX_{Q10}
-1.6E-01	0	-1.2E-01	0	0	-5.0E-01	-2.3E-01

$ZX_{t,ref}$	ZX_{init}	ZX_{max}	k_{ZX}	$maxAllo$	$allo$	$maxDens$
1.2E-01	0	3.7E-01	-3.3E-01	0	-5.8E-02	0

Table C.5: FORM (First Order Reliability Method) alpha values for the parameters used in the Monte Carlo analysis.



(a) A sample of 100 calculations in a Monte Carlo simulation. Each line represents biomass changes over time with different, randomly drawn values from the distributions as defined in Table C.4 using the equations as defined in Section C.5.



(b) Histogram of coral biomass after running the productivity model over a time period of approximately three years (model time, not calculation time). 10000 Monte Carlo samples were taken. The mean of the distribution is ≈ 16.8; limits of the 95% confidence interval are indicated by the vertical lines. The peak around 0 is probably caused by the fact that the answer is normally distributed, but the model prevents negative biomass values (which do not make physical sense); this causes all the otherwise negative values to accumulate in around the zero value.

Figure C.10: Results of a Monte Carlo simulation with no dredging induced excess TSS.

C.7 Findings

From the probabilistic calculation in Subsection C.6.3 and the descriptions of all the variables in Subsection C.6.1, it turns out there are a significant number of variables which have a large uncertainty, starting with the light extinction and carbon production equations which are reasonably well understood (see Appendix D for a more elaborate description), but rely on accurate measuring campaigns for obtaining both coral productivity parameters and local light extinction parameters. Assumptions such as a constant background coefficient (K_B) and a uniformly distributed TSS concentration need to be checked, and especially for the coral parameters it seems worthwhile to map the species present on a reef (and in what proportions). Appendix D showed that not only species can differ quite a bit, but genotypes (variations within a species) can cause differences as well. For light, local measurements (near the coral reef) could also remove quite some uncertainty.

The more coral specific relations have less ‘hard’ data; this is reflected in the choice for standard deviations expressing this uncertainty. The substrate limitation relation containing k_{ZX} deserves more research because the parameter choice is based on mimicking a different curve; it’s worthwhile finding out if the other curve is more suitable in a sense that it might have more data to calibrate on. Another relation that should be highlighted is the production allocation relation: during discussions with the author of Larsen (2012), the allocation (and thereby the allocation switch and parameters) are highly uncertain. Intuitively, it seems odd to divide the photosynthetic production first between zooxanthellae and coral host, and then subtract respiration costs. Because the zooxanthellae are only capable of creating photosynthetic products, it would make sense for them to first allocate a part of these products for respiration before handing the remainder of to the coral host. But, this a comment based on intuition and should be checked in literature to see if it holds any merit. Finally, there are a number of relations included which have some sort of temperature influence, but for most of them in Larsen (2012) this temperature dependency was only recognized qualitatively, but actually kept outside any calculations.

Based on this brief investigation focused primarily on the energy budget model described Larsen (2012), there are some significant gains to be made which will benefit not only this specific coral productivity model. Some of these items were already recognized in the recommendations of Larsen (2012). Not all of the recommendations of Larsen (2012) are repeated in order to keep the recommendations focused primarily on the results of the Monte Carlo calculation:

- Background TSS and light: seasonality, expected variations and depth variations are a number of points which would increase understanding. Mapping events such as algal bloom (which can significantly influence light transmission) is another issue which deserves attention.

- Viability of local light measurements instead of transforming subsurface light; or even transforming an entire light distribution to subsurface irradiation and to PAR; see also Appendix D.

- Temperature influences. Either the relations need to be investigated further, or a choice could be made to leave temperature dependence out altogether. The latter would require setting up an allowed envelope of temperature variation where significant changes are not expected, which would limit the applicability of the model but would also simplify the model.

- Investigate how good of a proxy carbon production is coral health. Production rates are strictly speaking only indicative of a potential for growth (Schutter, 2010), but seem to be a reasonably good indication of coral health.
- Because of the similarity between seagrass and zooxanthellae (both are considered to be primarily autotrophic), the symbiosis describing relations could be taken out for application to seagrass, potentially heavily simplifying the productivity model and cutting out a significant amount of uncertainty. However, it is very likely that seagrass has its own complexities. As the model has not been applied to seagrass in this document, this remains a preliminary recommendation and no definitive statement can be (or is) made.
- Depth of the coral is currently modeled as a stochastic variable. This could be taken out and replaced by a more detailed description of an applicable depth gradient.

In general, it has to be found out how good results from energy budget models (such as the one described in Larsen (2012)) are when compared to measurements/experiments. Ideas from other implementations as found in for example Anthony and Connolly (2004) or Kooijman (2010) might have valuable contributions in further improving the model.

Appendix D

Light attenuation and corals

Because light is seen as a major, if not the primary, influence regarding coral health, some additional attention is given in this appendix to the equations governing light in the coral productivity model as described in Appendix C.

D.1 Light and photosynthesis

Photosynthesis is, crudely put, about turning light energy into chemical energy. The energy of light is typically measured as a general radiation term covering both photon terms and energy terms; when it is incident on a surface, radiation is also called irradiance. In this thesis it is assumed that the sun is the sole provider of the incident light. Solar light comes in a distribution of different wavelengths, but most photosynthetic organisms can not use the full wave band. Generally, it is assumed that only the spectrum between 400nm and 700nm can be used; but even in that spectrum not each waveband can be used with the same efficiency. This spectrum is also called the Photosynthetically Active Radiation (PAR) (Alados and Alados-Arboledas, 1999).

If the photosynthetic rates are normalized by the quanta of light absorbed (relative quantum yield), a picture like in Figure D.1 can be drawn. It is however a common assumption that the efficiency in the PAR spectrum is the same for each wavelength; which is shown in Figure D.1 by the idealized response lines.

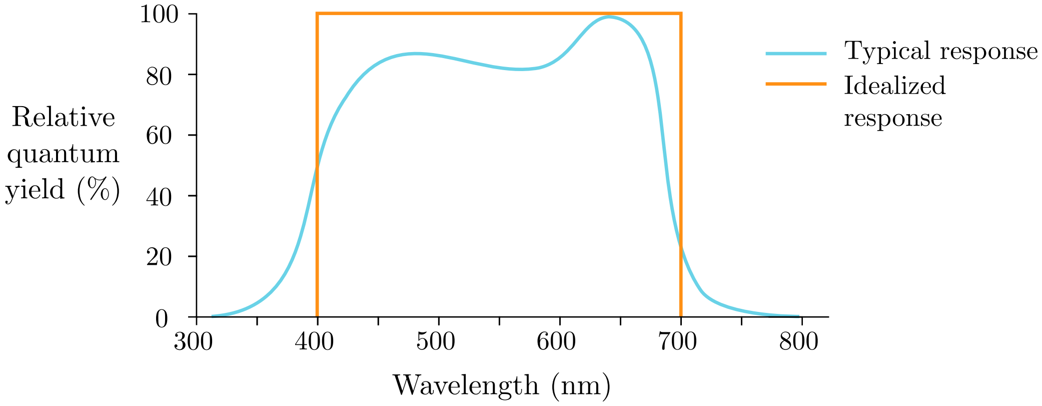


Figure D.1: Typical response and idealized quantum yield response. Image after "*Comparison of Quantum Sensors with Different Spectral Sensitivities*" by LI-COR.

Measuring PAR can either be done by using PAR sensors, or deriving it from broadband solar radiation. The latter requires conversions taking care of atmospheric conditions. Both methods need to account in some way for direct radiation and diffuse radiation (Alados and Alados-Arboledas, 1999). A commonly used unit for PAR is is the einstein unit (E), defined as one mole of photons regardless of frequency. Typically this unit is expressed per surface per time, e.g. $E/m^2/day$.

D.2 Light attenuation

Besides the PAR spectrum limiting the useful available light, the water column above the point of interest attenuates the irradiance. The content and optical properties of the water are important for this; TSS (Total Suspended Solids) is considered to be an important factor in the attenuation of irradiance. If the subsurface irradiation is measured, equation [C.3] shown in Section C.5 describes the transition from subsurface irradiation to local irradiation at the point of interest (in this case a coral reef). The subsurface irradiance is considered to be limited to the PAR spectrum only.

$$I(z) = I_0 \cdot e^{-(K_B + K_{TSS} \cdot TSS) \cdot z}$$

This is an implementation of the Beer-Lambert law for vertical attenuation of radiation:

$$I_{Z2} = I_{Z1} \cdot e^{-(K_d (Z_2 - Z_1)}$$

Where I_{Z1} and I_{Z2} are irradiances at depth Z_1 and Z_2; Z_2 is larger than Z_1 and K_d is the diffuse attenuation coefficient. If Z_2 is relative to Z_1, and Z_1 is set to zero, the equation simplifies to:

$$I_{Z2} = I_{Z1} \cdot e^{-K_d \cdot z} \qquad \text{[D.1]}$$

The diffuse attenuation coefficient K_d in [D.1] is significantly influenced by the absorption and scattering of light by water and dissolved or suspended substances. Xu et al. (2005) simplified the approximation of K_d for PAR by approximating and decomposing K_d in water, phytoplankton, seston (nonphytoplankton particulate matter) and chromophoric dissolved organic matter (CDOM):

$$K_d = K_{water} + K_{PHYTO} \cdot PHYTO + K_{SES} \cdot SES + K_{CDOM} \cdot CDOM \qquad \text{[D.2]}$$

In equation [D.2], TSS can be used as a proxy for seston, chlorophyll-a concentrations as a proxy for phytoplankton and salinity as a proxy for CDOM. Separating salinity and water attenuation was chosen in Xu et al. (2005) to separate uncertainty from CDOM and salinity interacting with each other; but the combination of water and CDOM is quite common as well:

$$K_d = K_{water+CDOM} + K_{PHYTO} \cdot PHYTO + K_{SES} \cdot SES \qquad \text{[D.3]}$$

During personal communication at DHI Singapore it was found that phytoplankton can be considered to be of negligible effect. This seems to be also found for a number of conditions in previous studies such as Borowiak (2005). This, in combination with using TSS as a proxy for seston, further reduces equation [D.3] for K_d. Combining this simplification with equation [D.1] leads to equation [C.3] in Section C.5:

$$\left.\begin{aligned} K_d &= K_{water+CDOM} + K_{SES} \cdot SES \\ K_d &= K_{water+CDOM} + K_{TSS} \cdot TSS \\ K_d &= K_B + K_{TSS} \cdot TSS \end{aligned}\right\} \qquad I_{Z2} = I_{Z1} \cdot e^{-K_d \cdot z} = I_{Z1} \cdot e^{-(K_B + K_{TSS} \cdot TSS) \cdot z}$$

In Larsen (2012) values for K_B and K_{TSS} were assumed to be 0.2649 and 0.063 respectively. In Xu et al. (2005) a value of 0.059 for K_{TSS} was found; even though this value was based on a location in Chesapeake Bay in the United States, it gives a rough estimate of the variability of the parameters in question. This variability can be largely attributed to the fact that light attenuation is modeled, which is not only dependent on the concentrations of for example TSS, but also on properties of the particles like their shape, color, and reflectivity (besides obvious suspects such as measuring accuracy/errors). These particle properties can differ significantly in space (and possibly in time as well); for more accurate results these parameters should be calibrated for local conditions. Interesting is that in Xu et al. (2005) was found that TSS could be considered as the dominant factor (for their dataset), reinforcing the hypothesis that TSS and light are significantly correlated.

D.2.1 Phytoplankton

Phytoplankton was chosen in the previous section to be a calibration value, but otherwise constant in time and depth averaged. This is not what is found in practice; algal 'blooms' are found which can significantly alter light attenuation. These blooms can be found to occur regularly, hinting that these might be correlated to seasonality or recurring flow patterns (upwelling for example). A numerical model which tries to capture the above and in general more facets of irradiance depth conversions is for example EcoLight (Mobley et al., 2009).

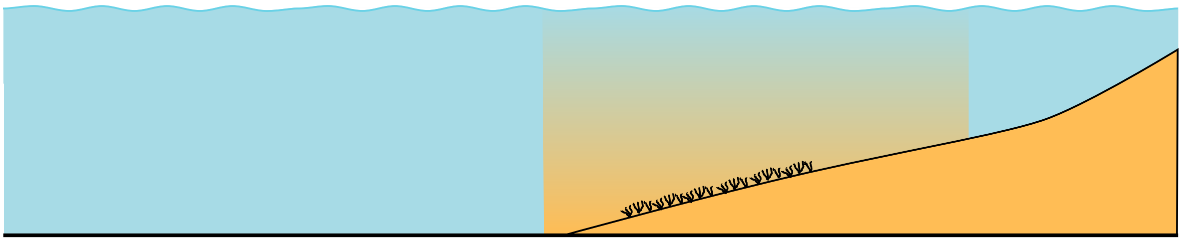
D.2.2 Influence depth and TSS on light attenuation

Equation [C.3] assumes a relation between TSS concentrations, depth and a background constant influencing the reduction of the subsurface irradiance. It is interesting to compare the reduction of the subsurface irradiance (I_0) using the previously mentioned values for K_B and K_{TSS} (which were set to 0.2649 and 0.063 respectively). This is done for a number of combinations of TSS and depth values in Table D.1.

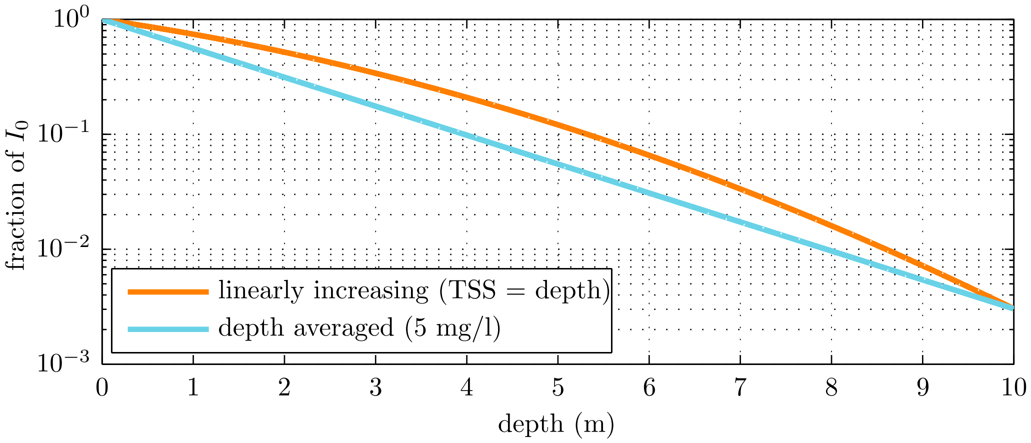
	TSS = 2 mg/l	**TSS = 5 mg/l**	**TSS = 10 mg/l**
z = 3m	$0.31 \cdot I_0$	$0.17 \cdot I_0$	$0.068 \cdot I_0$
z = 5m	$0.14 \cdot I_0$	$0.055 \cdot I_0$	$0.011 \cdot I_0$
z = 7m	$0.065 \cdot I_0$	$0.017 \cdot I_0$	$0.0019 \cdot I_0$

Table D.1: Influence of depth and TSS on the subsurface irradiance I_0.

Table Table D.1 shows that there is a significant influence of depth and TSS on the reduction of I_0. Changes in TSS are often reasonably to well modeled, but depth at a coral reef is often assumed as a single value when looking at excess dredging TSS impacts. If a coral reef has a changing depth profile, a lot of variability could be introduced if, for example, the reef varies between three and five meter depth. Even more interesting is a situation with a non-uniformly distributed TSS concentration over varying depth, which is depicted graphically in Figure D.2. In this figure a linearly increasing TSS concentration is drawn; results where TSS is equaled to the depth ($TSS = z$) are compared with a depth averaged TSS value in Figure D.2b.



(a) Fictitious coral reef on a slope, exposed to a non-uniformly distributed TSS concentration.



(b) Depth averaged, uniformly distributed TSS versus linearly increasing TSS.

Figure D.2: Influence of depth and non uniformly distributed TSS concentration on light extinction.

Figure D.2b shows that are significant differences regarding light extinction between a uniformly distributed TSS concentration profile, and a non-uniformly distributed TSS concentration profile, while a uniformly distributed TSS concentration profile is a commonly made assumption. These are all factors adding to uncertainty surrounding an answer, which make a probabilistic calculation (explicitly showing these uncertainties) necessary.

However, it should be said that the numbers in Table D.1 and Figure D.2 are strictly about light reduction. In Appendix C it is posed that (specifically in equations [C.4] and [C.5]) carbon production is dependent on the local light value; but given the nature of the relation (tangent hyperbolic) reductions in irradiance do not always have to significantly reduce production if the relation is near it's asymptotic value. However, this should be validated for each case before disregarding the influence of depth-variance.

D.3 Turbidity

Light attenuation and turbidity are closely related, but not the exactly the same. Strictly speaking the Beer-Lambert law used in the previous section describes absorption of light by particles, while turbidity often measures the scattering/reflection of light due to particles. But in practice the two are used interchangeably.

A way to express turbidity is by using NTU (Nephelometric Turbidity Units), which is a measure of turbidity from a calibrated nephelometer. This is an often used measuring unit, but is difficult to directly relate to TSS as expressed in the previous section. This is because a nephelometer measures the reflection of particles present in a fluid: contrary to relations such as found in [C.3] where attenuation is approximated based on TSS concentrations (and other factors). This means that relations found between TSS and NTU are most probably only valid for the conditions in which the relations were found. Several linear relations fitting NTU to TSS (e.g. Ziegler et al. (2011)) use factors in the order of 0.8-1.2: as a first order estimate NTU values could also be seen as TSS values in mg/l. It should be stressed that this is a rough approximation.

For the coral productivity model described in Appendix C, a constant background TSS value of 6 mg/l was chosen, based on a similar choice made in Larsen (2012). TSS, and turbidity, can not always be considered constant as was done in the coral productivity model by Larsen (2012) and in Appendix C. TSS, and turbidity, are likely to vary; for example in a river considerably higher turbidity values were found during peak discharges in Ziegler et al. (2011). Based on this, it is not too far fetched to assume for oceanic conditions that possibly wave climate, tides and ship wakes can significantly influence TSS (and/or turbidity) values.

D.4 Production curves

In Subsection D.2.2 mention was made that relative light availability cannot always be interpreted in a straight forward fashion: absolute light values are important in production curves. An example of a production curve can be found in Subsection C.5.1 in equation [C.6], which is repeated here:

$$
\begin{aligned}
P(I)_{nett} &= R_{dark} + P_{max} \cdot tanh\left(\frac{\alpha \cdot I(z)}{P_{max}}\right) \\
P(I)_{nett} &= R_{dark} + P_{max} \cdot tanh\left(\frac{I(z)}{I_k}\right)
\end{aligned}
$$

Where the angle α (in radians) in the second equation is equaled to P_{max}/I_k as opposed to $tan\,(P_{max}/I_k)$; this approximation is actually quite good as long as α is (much) smaller than one. For three coral reef locations, named CRA CRB and CRC, DHI Singapore performed a number of production-irradiance experiments by means of selecting a number of species of corals and transplanting these. After an acclimatisation period, samples were taken to a lab and measurements were made. Results of using a part of these measurements for fitting production curves by means of nonlinear curve-fitting in least-squares sense (using MATLAB) can be found in Table D.2 and Figure D.3.

Looking at the results, three point of views can be taken: looking at differences between species, differences between sites and differences between origins. In general, it seems like *Merulina* has a higher dark respiration but also a higher maximum production than *Pachyseris*. As far as differences between sites go, CRC seems to induce a higher dark respiration than CRA and CRB. This is in line with expectations for a high turbidity region (see also Section C.2.1).

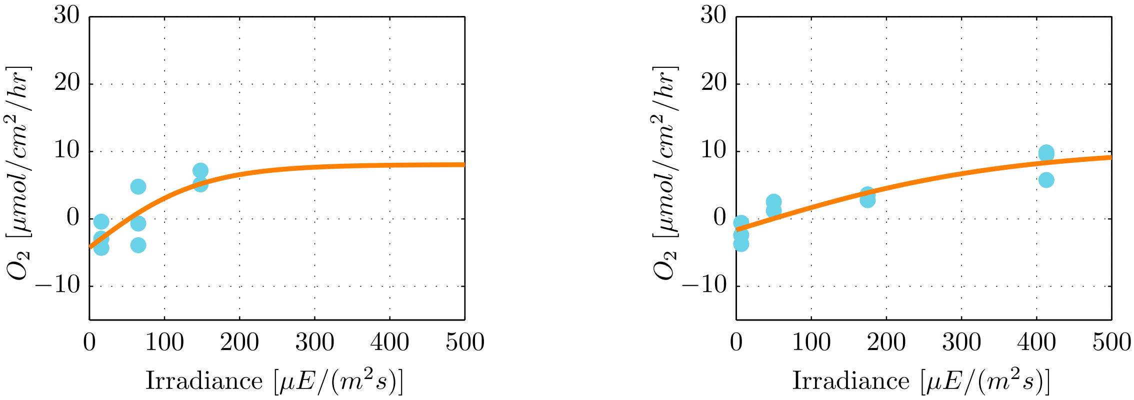
On the other hand, differences in origins are hard to distinguish, if there are any present. The results can be obscured by the variability in the data points with regard to the equation used for fitting, the (for this purpose) limited number of points per curve, or absence of differences at all. It should be noted that each each curve fit for a species contains measurements of about three different genotypes, each which could have slightly different parameters. Fitting of curves to each genotype has not been attempted because this would result in fitting curves to three or less data points per fit.

Furthermore, I_k varies significantly between fits: this can be explained by the variability in the data points and the fact that I_k is a point close to the asymptotic value of the tangent hyperbolic: small changes in the asymptotic value will lead to significant changes in I_k; for a graphical representation see Figure C.7 in Subsection C.5.1.

The main purpose of making the curve fits and analyzing the results is to gain a better insight in the variability of the parameters associated with the used production curve formulation (and the specific measuring method used by DHI Singapore). Based on these numbers, crude estimates for the standard deviation of R_{dark}, P_{max} and α have been set to 1.00, 1.00 and 0.0100 respectively. These estimates are used in Section C.6.

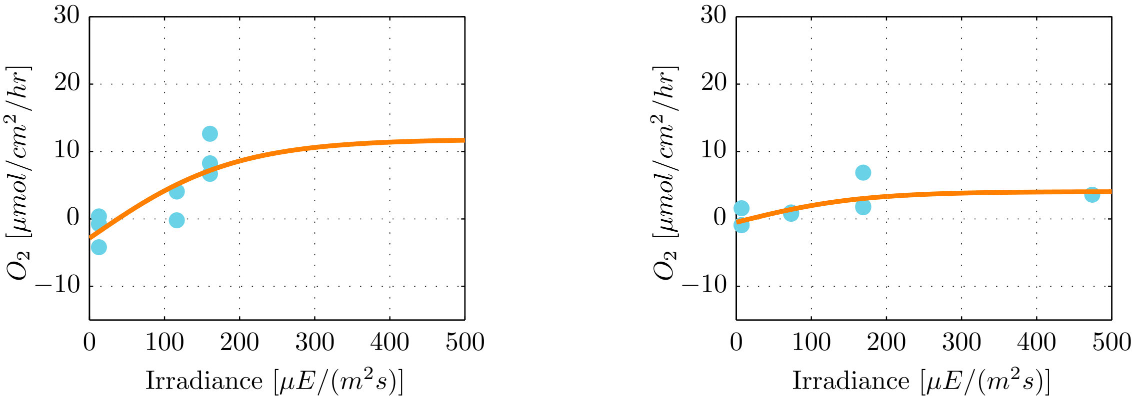
Site	**Origin**	**Species**	$\boldsymbol{R}_{dark}$	$\boldsymbol{P}_{max}$	$\boldsymbol{I}_k$	$\boldsymbol{R}^2$	$\boldsymbol{\alpha}$
CRA	CRA	Merulina	-4.24	12.32	146.29	*0.75*	8.4E-02
CRA	CRB	Merulina	-1.37	11.67	220.39	0.85	5.3E-02
CRA	CRC	Merulina	-0.65	5.86	167.22	0.86	3.5E-02
CRB	CRB	Merulina	-2.06	13.52	196.48	*0.72*	6.9E-02
CRB	CRA	Merulina	-2.85	14.69	191.93	*0.75*	7.7E-02
CRB	CRC	Merulina	-1.82	5.57	*60.46*	0.85	9.2E-02
CRC	CRC	Merulina	-3.12	9.83	160.80	0.87	6.1E-02
CRC	CRA	Merulina	-3.45	13.23	186.04	0.80	7.1E-02
CRC	CRB	Merulina	-2.80	12.41	245.26	0.88	5.1E-02
CRA	CRA	Pachyseris	-1.63	12.16	357.13	0.86	3.4E-02
CRA	CRB	Pachyseris	-0.78	30.54	*1632.49*	*0.76*	1.9E-02
CRA	CRC	Pachyseris	-1.67	8.09	179.25	0.87	4.5E-02
CRB	CRB	Pachyseris	-0.68	8.34	191.29	*0.72*	4.4E-02
CRB	CRA	Pachyseris	-0.49	4.55	163.94	*0.50*	2.8E-02
CRB	CRC	Pachyseris	*0.56*	4.43	357.35	*0.67*	1.2E-02
CRC	CRC	Pachyseris	-1.42	7.52	164.70	0.83	4.6E-02
CRC	CRB	Pachyseris	-1.46	6.75	*76.53*	0.84	8.8E-02
CRC	CRA	Pachyseris	-1.39	8.54	159.32	*0.66*	5.4E-02

Table D.2: Two species' production and irradiation parameters, grouped by the current location. Emphasized are values that seem off (have a value outside the perceived acceptable range for the parameter) or don't have a high R^2 value.



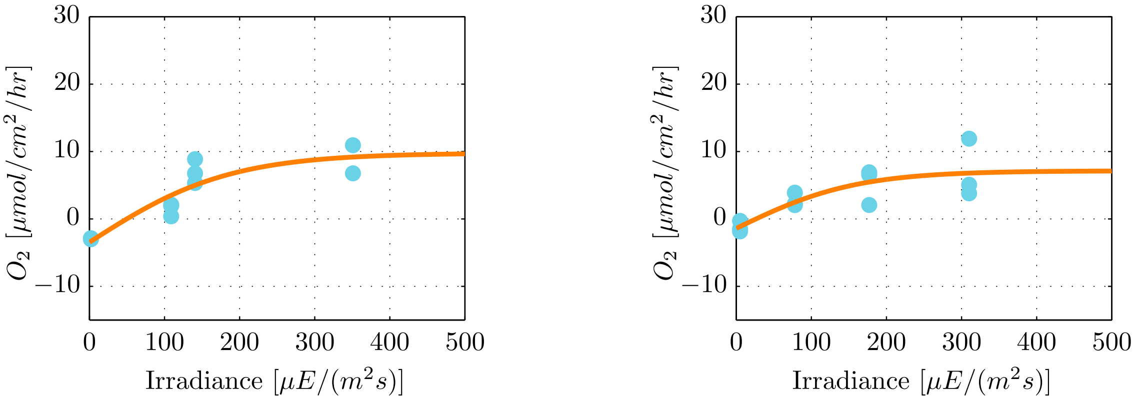
(a) Location: CRA, origin: CRA, species: Merulina

(b) Location: CRA, origin: CRA, species: Pachyseris



(c) Location: CRB, origin: CRA, species: Merulina

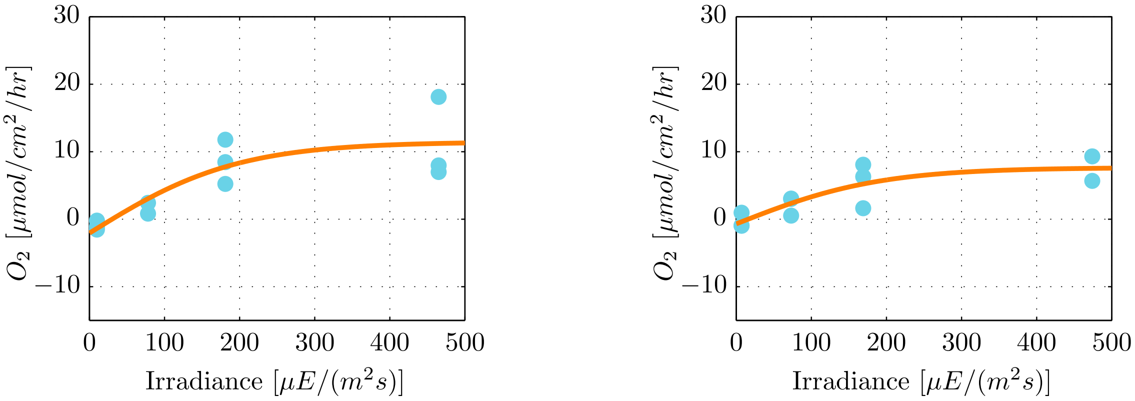
(d) Location: CRB, origin: CRA, species: Pachyseris



(e) Location: CRC, origin: CRA, species: Merulina

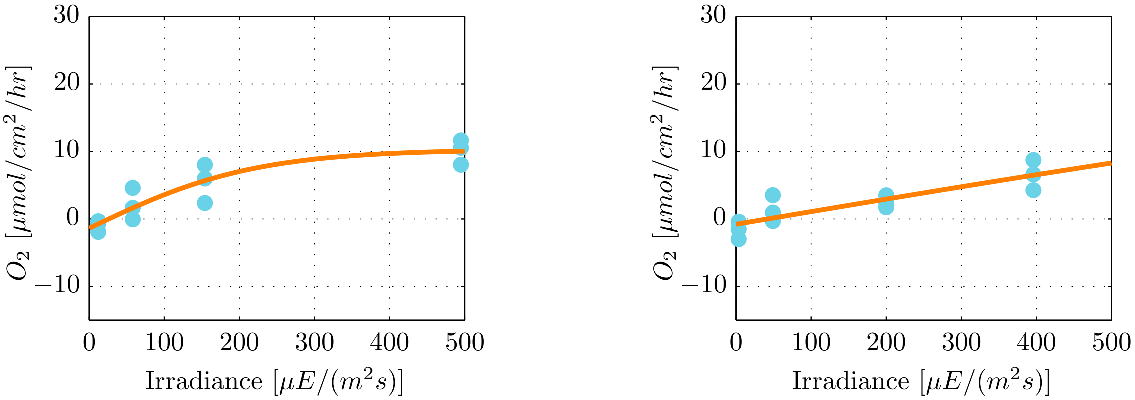
(f) Location: CRC, origin: CRA, species: Pachyseris

Figure D.3: Production/irradiance curves for two coral species. Data courtesy of DHI Singapore.



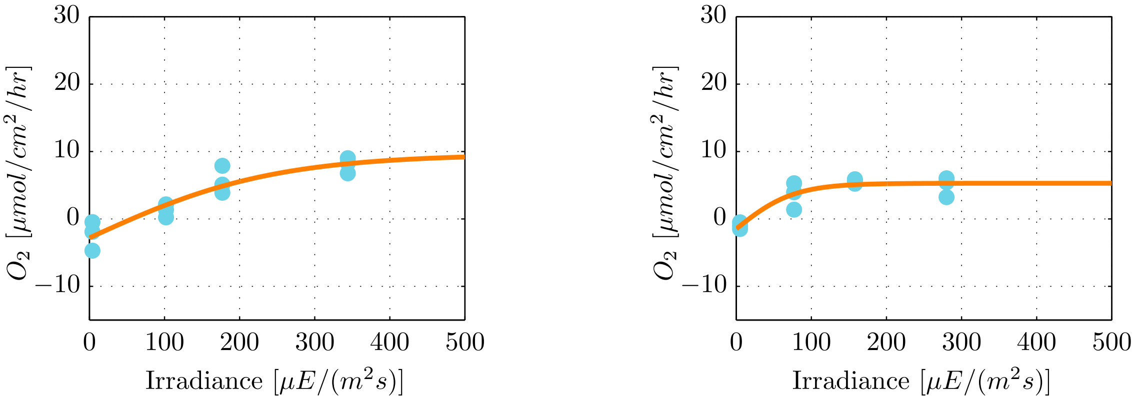
(g) Location: CRB, origin: CRB, species: Merulina

(h) Location: CRB, origin: CRB, species: Pachyseris



(i) Location: CRA, origin: CRB, species: Merulina

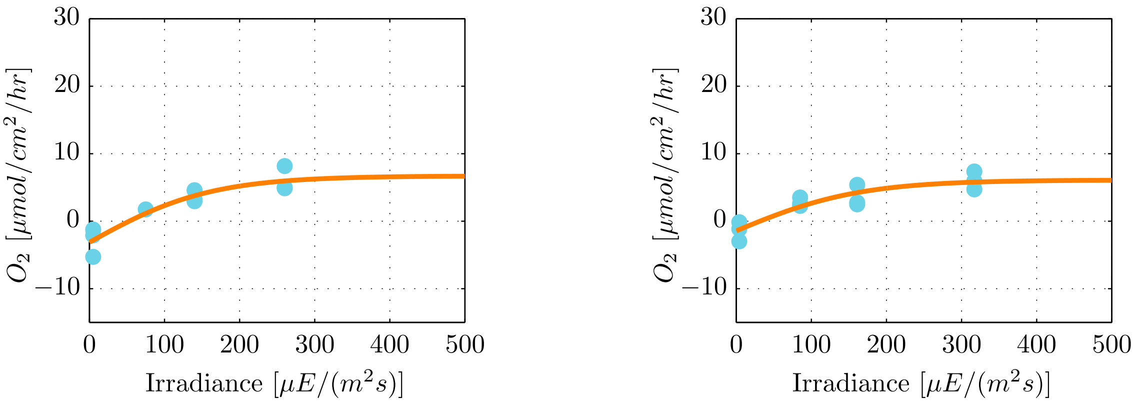
(j) Location: CRA, origin: CRB, species: Pachyseris



(k) Location: CRC, origin: CRB, species: Merulina

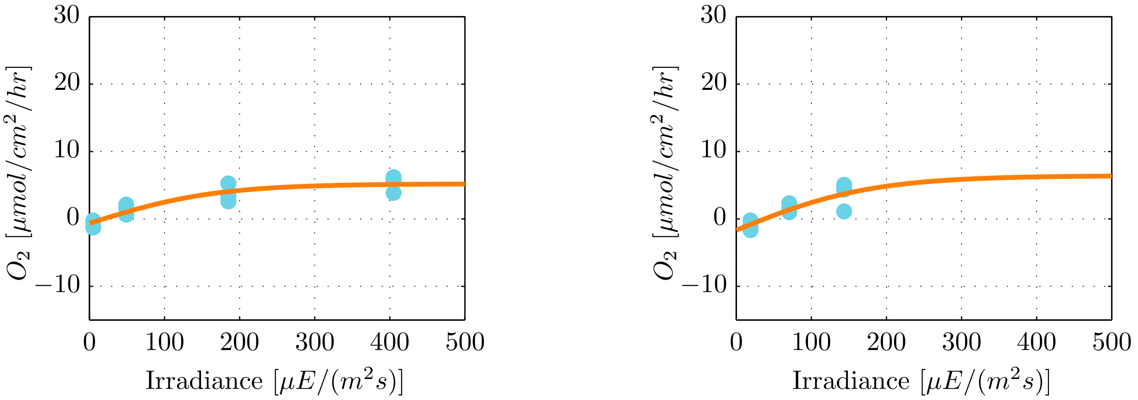
(l) Location: CRC, origin: CRB, species: Pachyseris

Figure D.3: (Continued) Production/irradiance curves for two coral species. Data courtesy of DHI Singapore.



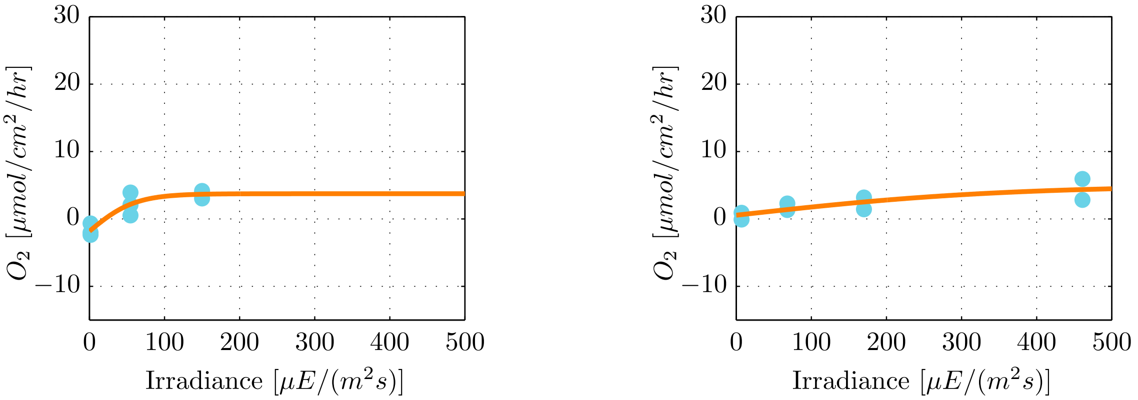
(m) Location: CRC, origin: CRC, species: Merulina

(n) Location: CRC, origin: CRC, species: Pachyseris



(o) Location: CRA, origin: CRC, species: Merulina

(p) Location: CRA, origin: CRC, species: Pachyseris



(q) Location: CRB, origin: CRC, species: Merulina

(r) Location: CRB, origin: CRC, species: Pachyseris

Figure D.3: (Continued) Production/irradiance curves for two coral species. Data courtesy of DHI Singapore.

Appendix E

Temperature and coral bleaching

E.1 Bleaching

Coral bleaching is the expelling of the symbiont zooxanthellae from the coral host due to stress. It has also been hypothesized that corals expel their zooxanthellae in order to replace them with a more stress-resistant zooxanthellae (Berkelmans and van Oppen, 2006); this does require the presence of more stress-resistant zooxanthellae in the vicinity of the coral and coral hosts which are capable of displaying this behavior. In general temperature is seen as the most prominent factor inducing bleaching, though other factors have been identified as well; for more information see Section C.2.1. For an quick overview of bleaching events, see Figure E.1b.

E.2 Temperature induced bleaching

In the case of temperature induced bleaching, two approaches are commonly used which differ in either considering acute (maximum short-term) heat stress or cumulative temperature stress (Baker et al., 2008). Both method types rely on accurately setting temperature thresholds. In practical applications, it turns out that a combination between acute heat stress (Podest and Glynn, 1997; Manzello et al., 2007), as well as cumulative temperature stress (Liu et al., 2003) together best predict bleaching, with prior temperature variability also increasing accuracy (McClanahan et al., 2007b,a; Baker et al., 2008). Data for hindcasting/calibration mainly comes from satellite measurements of Sea Surface Temperature (SST): commonly used databases are (to name a few) HadISST, ERSST and GISST (Baker et al., 2008).

Metrics used for acute heat stress could be the maximum absolute SST and/or the maximum SST anomaly; anomaly is defined as the difference between some threshold (commonly the mean temperature) and the actual temperature (Podest and Glynn, 1997). Another metric could be the so called “HotSpot”: it indicates the exceedance of the climatological mean temperature of the climatologically hottest month for a specific region (Liu et al., 2003).

For cumulative temperature stress the cumulative part is in the time period taken which can be days (Degree Days, DD), weeks (Degree Heating Weeks, DHW) or months (Degree Heating Months, DHM). For some definitions multiple models or definitions exist, but in this thesis only the most recent ones will be discussed:

Degree Days (DD) was introduced in Podest and Glynn (1997) and is calculated by first determining the "warm phase" of an area, followed by summing over the duration of the warm phase the differences between daily SST values and the SST overall mean. The warm phase is defined as the period(s) during the year where the SST is above the overall mean SST.

Degree Heating Weeks, (DHW) was defined by Liu et al. (2003) as the accumulation of HotSpots for a given location, over a rolling 12-week time period. However, HotSpots with values lower than 1°C are not included, as these are believed not to show any significant effects. To place this definition in context, one (1) DHW is the same as one week of HotSpot values staying at 1°C, or half a week at 2°C.

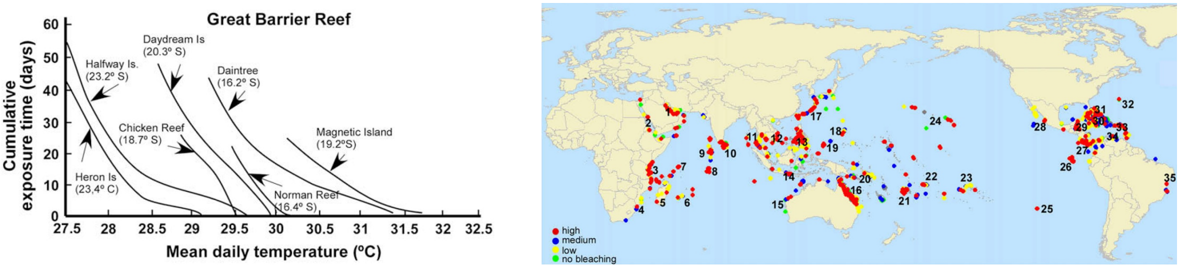
Degree Heating Months, (DHM) are similar to Degree Heating Weeks, but different in that monthly averages are used instead of weekly. In the implementation by Barton and Casey (2005), the reasoning for using monthly values was the desire for studying longer time periods. This also meant using data which had a lower spatial and temporal resolution; this had the consequence that extreme events were "averaged out" (damped). It was solved in Barton and Casey (2005) by dropping the 1°C threshold.

E.3 Limitations

These models are, however, very much high level models, both spatially and in the broad-stroke assumptions made; part of the variability (uncertainty) in the results found can be attributed to the precision of the dataset used, changing flow conditions and the degree of the affected reefs being "accustomed" to heat stress. High flow conditions generally decrease the severity of bleaching events by a two-fold mechanism: especially for shallow areas, local areas are better mixed resulting in potentially cooler areas, and the increased removal of oxygen radicals (see Section C.2.1). However, *because* of the just-mentioned reasons, corals which already managed to survive in low flow conditions might be better accustomed to heat anomalies compared to corals living in high flow conditions. The currently attributed capability for adaption lies mostly with the zooxanthellae because of their greater and faster capability to adapt relative to coral hosts (Császár et al., 2010). Besides horizontal flow components, vertical flow components (such as upwelling) might also significantly influence local results. It would be interesting to add, next to intensity and duration of temperature, a flow component to the analysis (or, in a more generally relevant approach, calculated local water temperature); having said that, upwelling has also been linked to aggravate bleaching due to influencing local flow patterns which ended up (rather paradoxically) increasing temperatures locally at coral reefs (Berkelmans R. and Steinberg, 2010). The impact of flow conditions on coral has been found to be significant; see for example Schutter (2010). To make it more general applicable, latitude seems to correlate with differences in bleaching sensitivity (other than the rather obvious species differences) (Berkelmans, 2002b). Given the problem set and the slightly varying but overall similar implementations, it would also be interesting to try and unify the approaches using a logit/probit approach for estimating bleaching impact. It should be noted that Berkelmans (2002b) was in possession of locally measured in-situ data, which is of higher quality than the average satellite measurement.

E.4 Possible additions

While these approaches are fairly straightforward, there are a number of assumptions made which strictly might not be true. First of all, the physical interpretation of a fixed temperature threshold is quite arbitrary: the threshold might change depending on governing conditions. In an attempt to fix this Berkelmans (2002b,a) introduced a threshold defined as $T_b = T_n$, which is the difference between the temperature distribution of the coolest bleaching year (T_b) and the warmest non-bleaching year (T_n), visualized in Figure E.1a. This also means days no longer need to be consecutive (however, day-to-day temperatures are highly auto-correlated, meaning consecutive blocks will still occur). Second of all, the definitions of cumulative temperature stress all implicitly assume a linear relation: duration x with intensity y is the same as duration y with intensity x. Strictly speaking, this is probably not true, but as an approximation it might hold for certain ranges of x and y (Manzello et al., 2007).



(a) Time-integrated bleaching threshold of Berkelmans (2002b) for reef sites on the Great Barrier Reef, Australia.

(b) Locations of bleaching reports (map from ReefBase, www.reefbase.org).

Figure E.1: Estimated bleaching threshold curves and actual reported bleaching spots. After Baker et al. (2008) and sources cited within.

Further fueling motivation towards looking at probit/logit analysis, is the finding of a linear relation on log-log scale in Winter et al. (1998). As this was also found in the application of a probit model in Becker (2011), this could be an indication of Haber's rule (Miller et al., 2000; Becker, 2011).

E.5 Mitigating factors

Mitigating factors for bleaching can include low light, high flow, lower temperatures and higher nutrients (Baker et al. (2008), and sources cited within) and are usually correlated with the habitat of coral reefs. Some could also be influenced by events which only happen occasionally, such as increased matter content or storms. Increased matter content can attenuate potentially harmful levels of irradiation. while storms can be associated with higher flow conditions or upwelling leading to possibly lower temperatures (Baker et al. (2008) and sources cited within). Higher matter content can be induced (for example) by dredging; the (passive) dredge plumes described in this report are elevated concentrations of fine sediment in the water column.

However, storms are often associated with significantly damaging coral reefs (Baker et al. (2008) and sources cited within); generally dredging operations are also associated with negative effects. In order for any of these to actually result in a net positive effect, bleaching needs to be imminent and detrimental effects of the event need to be small compared to the expected positive effects. This makes the above proposed positive (or ‘negative reducing’) effects preliminary until more evidence for these positive effects is found, and only probable for use in a limited set of conditions.

Appendix F

Analysis of DHI data

ID	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*
CR1	1	1E+10	0.0677	0.592	-0.18	0.307	2	1E+10	0.0465	0.713	-0.2	0.287	0.5	1E+10	0.0835	0.508	-0.16	0.321
CR1	1	**52**	-0.133	0.292	-0.36	0.115	2	**52**	-0.112	0.376	-0.35	0.136	0.5	52	-0.094	0.458	-0.33	0.154
CR1	1	24	-0.101	0.422	-0.34	0.146	2	24	-0.062	0.622	-0.3	0.184	0.5	**24**	-0.129	0.304	-0.36	0.118
CR1	1	12	-0.041	0.746	-0.28	0.205	2	12	-0.011	0.93	-0.25	0.233	0.5	12	-0.075	0.552	-0.31	0.172
CR1	1	4	-0.018	0.887	-0.26	0.227	2	4	-0.004	0.977	-0.25	0.24	0.5	4	-0.043	0.733	-0.28	0.203
CR1	1	2	0.0059	0.963	-0.24	0.249	2	2	0.001	0.993	-0.24	0.245	0.5	2	-0.018	0.887	-0.26	0.227
CR1	1	1	0.0439	0.728	-0.2	0.285	2	1	0.0169	0.894	-0.23	0.26	0.5	1	0.0184	0.884	-0.23	0.261
CR2	1	1E+10	-0.129	0.307	-0.36	0.119	2	1E+10	-0.146	0.246	-0.38	0.101	0.5	1E+10	-0.118	0.348	-0.35	0.129
CR2	1	**52**	-0.202	0.107	-0.42	0.045	2	**52**	-0.173	0.169	-0.4	0.074	0.5	**52**	-0.199	0.111	-0.42	0.047
CR2	1	24	-0.148	0.238	-0.38	0.099	2	24	-0.085	0.503	-0.32	0.163	0.5	24	-0.18	0.151	-0.41	0.066
CR2	1	12	-0.052	0.681	-0.29	0.194	2	12	-0.004	0.976	-0.25	0.24	0.5	12	-0.093	0.462	-0.33	0.154
CR2	1	4	-0.018	0.886	-0.26	0.227	2	4	0.0205	0.871	-0.22	0.263	0.5	4	-0.048	0.704	-0.29	0.198
CR2	1	2	-0.008	0.95	-0.25	0.236	2	2	0.0292	0.817	-0.22	0.271	0.5	2	-0.034	0.786	-0.28	0.211
CR2	1	1	0.0124	0.922	-0.23	0.256	2	1	0.02	0.874	-0.23	0.263	0.5	1	0.0072	0.955	-0.24	0.251

Table F.1: Correlation coefficient R (with confidence limits *Rlo* and *Rup*) and null hypothesis p for various dosefactors n and timeperiods *Period*. The timeperiod is given in weeks; if 1E+10 is given it means the entire previous period. **No time lag**. Emphasized values indicate highest correlation and lowest p per dosefactor n.

ID	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*
CR1	1	**1E+10**	0.187	0.135	-0.06	0.412	2	**1E+10**	0.186	0.138	-0.06	0.411	0.5	**1E+10**	0.139	0.268	-0.11	0.371
CR1	1	52	-0.12	0.325	-0.36	0.124	2	52	-0.13	0.319	-0.36	0.122	0.5	52	-0.11	0.368	-0.35	0.134
CR1	1	24	-0.03	0.8	-0.27	0.213	2	24	-0.03	0.793	-0.27	0.212	0.5	24	-0.02	0.868	-0.26	0.224
CR1	1	12	-0.01	0.963	-0.25	0.238	2	12	0.021	0.868	-0.22	0.264	0.5	12	-0.01	0.917	-0.26	0.231
CR1	1	4	0.027	0.828	-0.22	0.27	2	4	0.018	0.887	-0.23	0.261	0.5	4	0.021	0.867	-0.22	0.264
CR1	1	2	0.047	0.71	-0.2	0.288	2	2	0.039	0.757	-0.21	0.28	0.5	2	0.031	0.804	-0.21	0.273
CR1	1	1	0.074	0.56	-0.17	0.312	2	1	0.053	0.677	-0.19	0.293	0.5	1	0.05	0.69	-0.2	0.291
CR2	1	1E+10	0.015	0.904	-0.23	0.258	2	1E+10	-0.08	0.523	-0.32	0.166	0.5	1E+10	0.017	0.896	-0.23	0.259
CR2	1	**52**	-0.15	0.248	-0.38	0.102	2	**52**	-0.15	0.224	-0.38	0.095	0.5	**52**	-0.16	0.208	-0.39	0.089
CR2	1	24	-0.09	0.472	-0.33	0.157	2	24	-0.08	0.517	-0.32	0.165	0.5	24	-0.07	0.566	-0.31	0.174
CR2	1	12	-0.02	0.905	-0.26	0.23	2	12	0.001	0.994	-0.24	0.245	0.5	12	-0.02	0.856	-0.27	0.222
CR2	1	4	-0.01	0.915	-0.26	0.231	2	4	0.021	0.871	-0.22	0.263	0.5	4	-0.04	0.726	-0.29	0.202
CR2	1	2	-0.01	0.961	-0.25	0.238	2	2	0.029	0.821	-0.22	0.271	0.5	2	-0.03	0.811	-0.27	0.215
CR2	1	1	0.012	0.925	-0.23	0.255	2	1	0.019	0.878	-0.23	0.262	0.5	1	0.004	0.975	-0.24	0.248

Table F.2: Correlation coefficient R (with confidence limits *Rlo* and *Rup*) and null hypothesis p for various dosefactors n and timeperiods *Period*. The timeperiod is given in weeks; if 1E+10 is given it means the entire previous period. **No time lag, exp(0.01TSS)**. Emphasized values indicate highest correlation and lowest p per dosefactor n.

ID	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*
CR1	1	1E+10	0.0666	0.598	-0.18	0.306	2	1E+10	0.0435	0.731	-0.2	0.284	0.5	1E+10	0.0832	0.51	-0.16	0.321
CR1	1	**52**	-0.134	0.287	-0.37	0.114	2	**52**	-0.114	0.364	-0.35	0.133	0.5	52	-0.092	0.467	-0.33	0.156
CR1	1	24	-0.108	0.392	-0.34	0.14	2	24	-0.068	0.592	-0.31	0.179	0.5	**24**	-0.134	0.286	-0.37	0.113
CR1	1	12	-0.052	0.684	-0.29	0.195	2	12	-0.019	0.878	-0.26	0.226	0.5	12	-0.085	0.499	-0.32	0.162
CR1	1	4	-0.031	0.803	-0.27	0.214	2	4	-0.025	0.841	-0.27	0.22	0.5	4	-0.047	0.712	-0.29	0.199
CR1	1	2	-0.028	0.825	-0.27	0.217	2	2	-0.049	0.698	-0.29	0.197	0.5	2	-0.032	0.799	-0.27	0.213
CR1	1	1	-0.013	0.917	-0.26	0.231	2	1	-0.067	0.597	-0.31	0.18	0.5	1	0.0013	0.992	-0.24	0.245
CR2	1	1E+10	-0.129	0.306	-0.36	0.119	2	1E+10	-0.146	0.245	-0.38	0.101	0.5	1E+10	-0.118	0.348	-0.35	0.129
CR2	1	**52**	-0.203	0.105	-0.43	0.043	2	**52**	-0.175	0.164	-0.4	0.072	0.5	**52**	-0.2	0.11	-0.42	0.046
CR2	1	24	-0.152	0.227	-0.38	0.096	2	24	-0.087	0.489	-0.32	0.16	0.5	24	-0.182	0.146	-0.41	0.064
CR2	1	12	-0.058	0.646	-0.3	0.188	2	12	-0.007	0.955	-0.25	0.237	0.5	12	-0.099	0.432	-0.33	0.148
CR2	1	4	-0.026	0.839	-0.27	0.22	2	4	0.0106	0.933	-0.23	0.254	0.5	4	-0.052	0.679	-0.29	0.194
CR2	1	2	0.0001	0.999	-0.24	0.244	2	2	0.0443	0.726	-0.2	0.285	0.5	2	-0.029	0.817	-0.27	0.216
CR2	1	1	-0.062	0.626	-0.3	0.185	2	1	-0.003	0.982	-0.25	0.241	0.5	1	-0.078	0.535	-0.32	0.169

Table F.3: Correlation coefficient R (with confidence limits *Rlo* and *Rup*) and null hypothesis p for various dosefactors n and timeperiods *Period*. The timeperiod is given in weeks; if 1E+10 is given it means the entire previous period. **Three day time lag**. Emphasized values indicate highest correlation and lowest p per dosefactor n.

ID	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*
CR1	1	1E+10	0.066	0.601	-0.18	0.305	2	1E+10	0.0445	0.725	-0.2	0.285	0.5	1E+10	0.0828	0.512	-0.16	0.32
CR1	1	**52**	-0.13	0.301	-0.36	0.117	2	**52**	-0.105	0.406	-0.34	0.143	0.5	52	-0.088	0.488	-0.32	0.16
CR1	1	24	-0.114	0.368	-0.35	0.134	2	24	-0.069	0.587	-0.31	0.178	0.5	**24**	-0.139	0.269	-0.37	0.109
CR1	1	12	-0.062	0.626	-0.3	0.185	2	12	-0.019	0.883	-0.26	0.226	0.5	12	-0.097	0.441	-0.33	0.15
CR1	1	4	-0.048	0.702	-0.29	0.198	2	4	-0.014	0.912	-0.26	0.231	0.5	4	-0.076	0.545	-0.31	0.171
CR1	1	2	-0.027	0.833	-0.27	0.219	2	2	-0.008	0.948	-0.25	0.236	0.5	2	-0.036	0.775	-0.28	0.21
CR1	1	1	-0.057	0.652	-0.3	0.19	2	1	-0.03	0.812	-0.27	0.215	0.5	1	-0.07	0.579	-0.31	0.177
CR2	1	1E+10	-0.129	0.306	-0.36	0.119	2	1E+10	-0.146	0.245	-0.38	0.101	0.5	1E+10	-0.118	0.348	-0.35	0.129
CR2	1	**52**	-0.204	0.103	-0.43	0.042	2	**52**	-0.177	0.159	-0.4	0.07	0.5	**52**	-0.2	0.11	-0.42	0.046
CR2	1	24	-0.153	0.224	-0.38	0.095	2	24	-0.087	0.492	-0.32	0.161	0.5	24	-0.183	0.144	-0.41	0.063
CR2	1	12	-0.062	0.623	-0.3	0.184	2	12	-0.009	0.946	-0.25	0.236	0.5	12	-0.105	0.405	-0.34	0.142
CR2	1	4	-0.021	0.866	-0.26	0.224	2	4	0.0028	0.982	-0.24	0.247	0.5	4	-0.043	0.731	-0.28	0.203
CR2	1	2	-0.009	0.944	-0.25	0.235	2	2	0.0263	0.836	-0.22	0.268	0.5	2	-0.034	0.791	-0.28	0.212
CR2	1	1	-0.026	0.835	-0.27	0.219	2	1	0.0308	0.808	-0.21	0.273	0.5	1	-0.065	0.606	-0.3	0.182

Table F.4: Correlation coefficient R (with confidence limits Rlo and Rup) and null hypothesis p for various dosefactors n and timeperiods *Period*. The timeperiod is given in weeks; if 1E+10 is given it means the entire previous period. **One week time lag**. Emphasized values indicate highest correlation and lowest p per dosefactor n.

ID	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*
CR1	1	1E+10	0.0661	0.601	-0.18	0.305	2	1E+10	0.0445	0.725	-0.2	0.285	0.5	1E+10	0.0829	0.511	-0.16	0.32
CR1	1	**52**	-0.122	0.331	-0.36	0.125	2	**52**	-0.101	0.422	-0.34	0.146	0.5	52	-0.078	0.538	-0.32	0.169
CR1	1	24	-0.119	0.347	-0.35	0.129	2	24	-0.072	0.57	-0.31	0.175	0.5	**24**	-0.142	0.259	-0.37	0.106
CR1	1	12	-0.062	0.623	-0.3	0.185	2	12	-0.021	0.868	-0.26	0.224	0.5	12	-0.099	0.432	-0.34	0.148
CR1	1	4	-0.047	0.712	-0.29	0.199	2	4	-0.008	0.952	-0.25	0.237	0.5	4	-0.087	0.489	-0.32	0.16
CR1	1	2	-0.042	0.742	-0.28	0.204	2	2	-0.007	0.955	-0.25	0.237	0.5	2	-0.063	0.616	-0.3	0.183
CR1	1	1	-1E-04	0.999	-0.24	0.244	2	1	0.0052	0.967	-0.24	0.249	0.5	1	-0.007	0.955	-0.25	0.237
CR2	1	1E+10	-0.128	0.309	-0.36	0.12	2	1E+10	-0.146	0.245	-0.38	0.101	0.5	1E+10	-0.118	0.351	-0.35	0.13
CR2	1	**52**	-0.207	0.098	-0.43	0.039	2	**52**	-0.184	0.142	-0.41	0.063	0.5	**52**	-0.2	0.11	-0.42	0.046
CR2	1	24	-0.159	0.205	-0.39	0.088	2	24	-0.097	0.442	-0.33	0.15	0.5	24	-0.185	0.139	-0.41	0.061
CR2	1	12	-0.063	0.618	-0.3	0.184	2	12	0.0022	0.986	-0.24	0.246	0.5	12	-0.11	0.383	-0.34	0.138
CR2	1	4	-0.054	0.671	-0.29	0.193	2	4	-0.024	0.847	-0.27	0.221	0.5	4	-0.071	0.574	-0.31	0.176
CR2	1	2	-0.036	0.776	-0.28	0.21	2	2	0.0034	0.978	-0.24	0.247	0.5	2	-0.064	0.611	-0.3	0.183
CR2	1	1	0.0189	0.882	-0.23	0.262	2	1	0.013	0.918	-0.23	0.256	0.5	1	0.0289	0.819	-0.22	0.271

Table F.5: Correlation coefficient R (with confidence limits *Rlo* and *Rup*) and null hypothesis p for various dosefactors n and timeperiods *Period*. The timeperiod is given in weeks; if 1E+10 is given it means the entire previous period. **Two week time lag**. Emphasized values indicate highest correlation and lowest p per dosefactor n.

ID	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*
CR1	1	1E+10	0.0658	0.603	-0.18	0.305	2	1E+10	0.0425	0.737	-0.2	0.283	0.5	1E+10	0.083	0.511	-0.16	0.32
CR1	1	52	-0.109	0.388	-0.34	0.139	2	**52**	-0.09	0.475	-0.33	0.157	0.5	52	-0.062	0.621	-0.3	0.184
CR1	1	**24**	-0.12	0.34	-0.35	0.127	2	24	-0.078	0.538	-0.32	0.169	0.5	**24**	-0.134	0.287	-0.37	0.114
CR1	1	12	-0.067	0.596	-0.31	0.18	2	12	-0.028	0.823	-0.27	0.217	0.5	12	-0.098	0.435	-0.33	0.149
CR1	1	4	-0.033	0.794	-0.27	0.213	2	4	-1E-04	0.999	-0.24	0.244	0.5	4	-0.073	0.565	-0.31	0.174
CR1	1	2	-0.049	0.698	-0.29	0.197	2	2	-0.008	0.951	-0.25	0.237	0.5	2	-0.099	0.432	-0.34	0.148
CR1	1	1	-0.04	0.754	-0.28	0.206	2	1	-0.011	0.929	-0.25	0.233	0.5	1	-0.077	0.541	-0.32	0.17
CR2	1	1E+10	-0.127	0.314	-0.36	0.121	2	1E+10	-0.145	0.249	-0.38	0.103	0.5	1E+10	-0.117	0.354	-0.35	0.131
CR2	1	**52**	-0.207	0.099	-0.43	0.039	2	**52**	-0.188	0.133	-0.41	0.058	0.5	**52**	-0.198	0.115	-0.42	0.049
CR2	1	24	-0.158	0.209	-0.39	0.089	2	24	-0.095	0.452	-0.33	0.152	0.5	24	-0.182	0.148	-0.41	0.065
CR2	1	12	-0.077	0.543	-0.31	0.17	2	12	-0.021	0.871	-0.26	0.225	0.5	12	-0.117	0.355	-0.35	0.131
CR2	1	4	-0.06	0.636	-0.3	0.187	2	4	-0.03	0.813	-0.27	0.216	0.5	4	-0.076	0.55	-0.31	0.172
CR2	1	2	-0.056	0.658	-0.3	0.191	2	2	-0.034	0.785	-0.28	0.211	0.5	2	-0.067	0.595	-0.31	0.18
CR2	1	1	-0.003	0.98	-0.25	0.241	2	1	-0.012	0.922	-0.26	0.232	0.5	1	0.0091	0.943	-0.24	0.252

Table F.6: Correlation coefficient R (with confidence limits *Rlo* and *Rup*) and null hypothesis p for various dosefactors n and timeperiods *Period*. The timeperiod is given in weeks; if 1E+10 is given it means the entire previous period. **Four week time lag**. Emphasized values indicate highest correlation and lowest p per dosefactor n.

ID	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*	*n*	*Period*	*R*	*p*	*Rlo*	*Rup*
CR1	1	1E+10	0.0645	0.61	-0.18	0.304	2	1E+10	0.0368	0.771	-0.21	0.278	0.5	1E+10	0.0833	0.509	-0.16	0.321
CR1	1	52	-0.089	0.483	-0.33	0.159	2	52	-0.086	0.493	-0.32	0.161	0.5	52	-0.038	0.763	-0.28	0.208
CR1	1	**24**	-0.121	0.338	-0.35	0.127	2	**24**	-0.093	0.463	-0.33	0.155	0.5	24	-0.107	0.395	-0.34	0.14
CR1	1	12	-0.099	0.434	-0.33	0.149	2	12	-0.071	0.571	-0.31	0.175	0.5	**12**	-0.111	0.38	-0.35	0.137
CR1	1	4	-0.063	0.616	-0.3	0.183	2	4	-0.036	0.776	-0.28	0.21	0.5	4	-0.089	0.483	-0.33	0.159
CR1	1	2	-0.02	0.873	-0.26	0.225	2	2	-0.017	0.895	-0.26	0.228	0.5	2	-0.038	0.765	-0.28	0.208
CR1	1	1	0.0188	0.882	-0.23	0.262	2	1	0.0146	0.908	-0.23	0.258	0.5	1	0.0059	0.963	-0.24	0.249
CR2	1	1E+10	-0.124	0.326	-0.36	0.124	2	1E+10	-0.142	0.259	-0.37	0.106	0.5	1E+10	-0.115	0.362	-0.35	0.133
CR2	1	**52**	-0.207	0.098	-0.43	0.039	2	**52**	-0.205	0.102	-0.43	0.041	0.5	**52**	-0.192	0.126	-0.42	0.055
CR2	1	24	-0.173	0.167	-0.4	0.074	2	24	-0.116	0.355	-0.35	0.131	0.5	24	-0.184	0.142	-0.41	0.063
CR2	1	12	-0.137	0.277	-0.37	0.111	2	12	-0.077	0.541	-0.32	0.17	0.5	12	-0.155	0.219	-0.38	0.093
CR2	1	4	-0.052	0.68	-0.29	0.194	2	4	0.0073	0.954	-0.24	0.251	0.5	4	-0.098	0.436	-0.33	0.149
CR2	1	2	-0.021	0.87	-0.26	0.224	2	2	0.0599	0.635	-0.19	0.299	0.5	2	-0.075	0.552	-0.31	0.172
CR2	1	1	-0.034	0.786	-0.28	0.211	2	1	-0.007	0.957	-0.25	0.237	0.5	1	-0.066	0.602	-0.3	0.181

Table F.7: Correlation coefficient R (with confidence limits *Rlo* and *Rup*) and null hypothesis p for various dosefactors n and timeperiods *Period*. The timeperiod is given in weeks; if 1E+10 is given it means the entire previous period. **Eight week time lag**. Emphasized values indicate highest correlation and lowest p per dosefactor n.