Chapter 10
Eutrophication

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Abstract Anthropogenic increases in nitrogen and phosphorus inputs to terrestrial and aquatic ecosystems have driven increases in eutrophication, the occurrence of ecosystem changes due to over-supply of nutrients. Eutrophic water bodies exhibit changes in species composition that often include algal blooms and oxygen depletion, with occasionally arresting images of fish kills or dead zones. Though dramatic and subtle consequences of eutrophication itself have been described for over 100 years, understanding of nutrients as the main drivers for this phenomenon is more recent. Modelling nutrient fate has reached a basic level of operability, with a general rule that freshwaters are limited in phosphorus (and hence respond to its addition), and terrestrial and marine systems are nitrogen limited. However, understanding of ecosystems responses such as species shifts or changes in primary productivity is still growing. Future work should incorporate more comprehensive metrics to quantify impacts of eutrophication on ecosystems – and the human systems that depend on them.

Keywords Freshwater eutrophication • LCA • LCIA • Life cycle assessment • Life cycle impact assessment marine eutrophication • Nutrient enrichment • Terrestrial eutrophication

1 Introduction

1.1 Historical Perspective

Eutrophication is the result of supplying nutrients to ecosystems in excess of natural rates, which may drive a cascade of changes, including alterations in species composition, biomass, or productivity. Nutrient cycling varies across ecosystems, and eutrophication does naturally occur to some degree, but anthropogenic

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emissions of nutrients and organic matter have increased nutrient cycling, disturbing the natural dynamic (Bouwman et al. 2009). This increased cycling, and often over-supply, of nutrients can directly affect humans: fish kills or red tides in aquatic systems may have direct health effects, and changing crop yields in terrestrial systems affects food production. Algal blooms continue to be of concern in many areas (e.g., Liu et al. 2013). Comprehensive overviews of nutrient-related ecosystem alterations are provided by Schindler (2006) and Smith et al. (1999).

The response of terrestrial systems to nutrient supply was documented in the mid-nineteenth century (von Liebig 1855). Excess nutrients have been a concern for over a century; a qualitative description of lakes as eutrophied, based on hypolimnetic oxygen depletion, the changed occurrence of benthic macroinvertebrates, and visual appearance of water, was first published in the early 1900s (Weber 1907). However, attention intensified in the middle of the last century (NRC 1992; OECD 1982), largely due to increased nitrogenous atmospheric emissions and synthetic fertilizer use, and the corresponding increase in the visibility of eutrophication and its impacts (MEA 2005; Tilman 1999; Vitousek et al. 1997). There was uncertainty about the causes of aquatic eutrophication into the 1960s, with a variety of substances, including phosphorus (P) and nitrogen (N), carbon, vitamins, amino acids, and trace elements identified as possible causes (NAS 1969). Even after phosphorus was identified as a critical substance in eutrophication (Vollenweider 1968), it took time – and demonstration via whole-lake manipulation – to overcome counter-claims and resistance from the scientific community and the soap and detergent industries (Schindler 2006).

Initial efforts to incorporate eutrophication into LCIA were largely based on biomass production (Lindfors et al. 1995); for aquatic eutrophication, this was connected to nutrient input via the Redfield ratio, the average stoichiometric ratios of carbon, nitrogen, and phosphorus(C_{106}:N_{16}:P_1) found in plankton (Redfield 1934). This ratio indicates that algae and other aquatic organisms require 16 mol of N for every one mole of P. A lack of one of these nutrients can limit biomass production; such systems are said to be N- or P-limited. A SETAC working group on LCIA noted that nearly all LCIA aquatic eutrophication methods based their characterisation factors on the Redfield ratio, with the corresponding impact indicator being algal growth (Udo de Haes et al. 2002). Alternately, some methods considered oxygen depletion in water, which has the advantage of providing a direct way to include organic matter, linking these to oxygen depletion via the biological or chemical oxygen demand (BOD or COD) (e.g., Guinée et al. 2002). The oxygen-consuming degradation of organic matter by bacteria forms the basis for the BOD; COD is a measure of all substances that can be oxidised. Newer LCIA methods have begun to move beyond the Redfield ratio and algae to consider changes in species composition, which itself is an interim step in the progression towards more complete assessments.

2   Principles of Characterisation Modelling

Modelling eutrophication requires capturing interrelationships between hydrology, ecosystem biology, microbiology, and chemistry. Modelling choices reflect differences both in scientific opinion about the cause-effect chain and in societal values about what merits protection (Hertwich et al. 2000; Udo de Haes et al. 2002; UNEP/SETAC 2005). Capturing complexity while creating a usable approach is a thread that runs through eutrophication modelling in LCIA.

One modelling choice regards the use of average or marginal impacts: if an LCIA is being conducted to assess a change or comparison, then marginal modelling is recommended; average impacts are suitable for information-gathering (Udo de Haes et al. 2002). To date, LCIA models for eutrophication have relied on marginal changes; e.g., Struijs et al. (2010a) used a 1 % increase above 1995 emission levels. However, this conceptual framework does discount impacts when the marginal increase occurs in an already-stressed receiving area (Huijbregts et al. 2011). Another modelling challenge, for both the average and marginal approaches, is the possibility of non-linear responses to nutrient loading. Loading may reduce ecosystem resilience; beyond critical levels, ecosystems may undergo radical shifts to alternate, metastable states (Scheffer et al. 2001).

2.1   Criteria for Good Characterisation Models

The variety of modelling approaches for eutrophication and other categories in LCIA has led to ongoing efforts to create a framework to objectively compare models (e.g., Udo de Haes et al. 2002; Margni et al. 2008). The most comprehensive effort to date has been the International Reference Life Cycle Data System (ILCD) (EC-JRC 2010a). In the ILCD framework, scientifically sound LCIA eutrophication models include a complete of scope, environmental relevance, and scientific robustness. These criteria require accurately capturing the cause-effect chain, to the extent made possible by current knowledge. For aquatic transport of eutrophying substances, major transport phenomena include precipitation and sedimentation for N and P, as well as oxidation, specifically denitrification, for N. For atmospheric emissions, these include oxidation and deposition. At the damage level, sound models should include discrimination between receiving areas based on sensitivity, possibly including a critical level, and a dose-response relationship (EC-JRC 2011). The application of these criteria to eutrophication models is discussed in the following sections.

3 Impact Pathway and Affected Areas of Protection

Damage categories represent changes to those components of the environment that are valued by human society (UNEP/SETAC 2005); however, valuing and quantifying ecosystem qualities is challenging, as society may value components ranging from the subjective (e.g., aesthetic quality) to the concrete (e.g., fish production for food supply). The Natural Environment Area of Protection, as defined by the UNEP/SETAC working group, encompasses both ends of this spectrum. The intrinsic values of the existence and stability of the environment is captured via biodiversity, which can be measured as a species loss. The functional values of natural resources are captured via net primary productivity, which can be measured financially (Margni et al. 2008). Some methods have considered human health, since direct human impacts are possible: algal blooms may be toxic to humans and have resulted in beach closures (Anderson 1989; Paerl et al. 2001). However, human health has rarely been considered directly for eutrophication in LCIA. Developing a set of metrics to fully capture the myriad aspects of the biotic environment is ongoing (see Sect. 8).

The addition of nutrients or organic matter to terrestrial or aquatic ecosystems can affect cell synthesis or energy supply. Organisms that can take advantage of changing inputs of either nutrients or organic matter will be able to outcompete other species, resulting in dynamic changes to the steady-state ecosystem composition, changing biodiversity and productivity. Figure 10.1 provides an overview of the terrestrial and aquatic eutrophication processes, which are discussed below.

3.1 Terrestrial Eutrophication

To date, LCIA has focused on changes to terrestrial vegetation, as interactions of other components of the ecosystem with changing nutrient cycling and changing plant communities are not yet well understood. Plants in terrestrial systems are usually nitrogen limited; i.e., there is sufficient P for growth, but not adequate N for the typical cellular nutrient ratio of N and P (see Sect. 1.1) (Grouzet et al. 2000; Hornung et al. 1994; Nilsson and Grennfelt 1988). The terrestrial N:P nutrient ratio, as well as the freshwater ratio, are similar to the Redfield ratio, although there are differences in the ratio of carbon to these nutrients (Elser et al. 2000; McGroddy et al. 2004). Excess nitrogen can change the structure and function of terrestrial, N-limited ecosystems by favoring a (typically) limited number of N-adapted species. This may change tolerance to disease or other stressors (e.g., drought, frost), resulting in overall biodiversity and productivity changes.
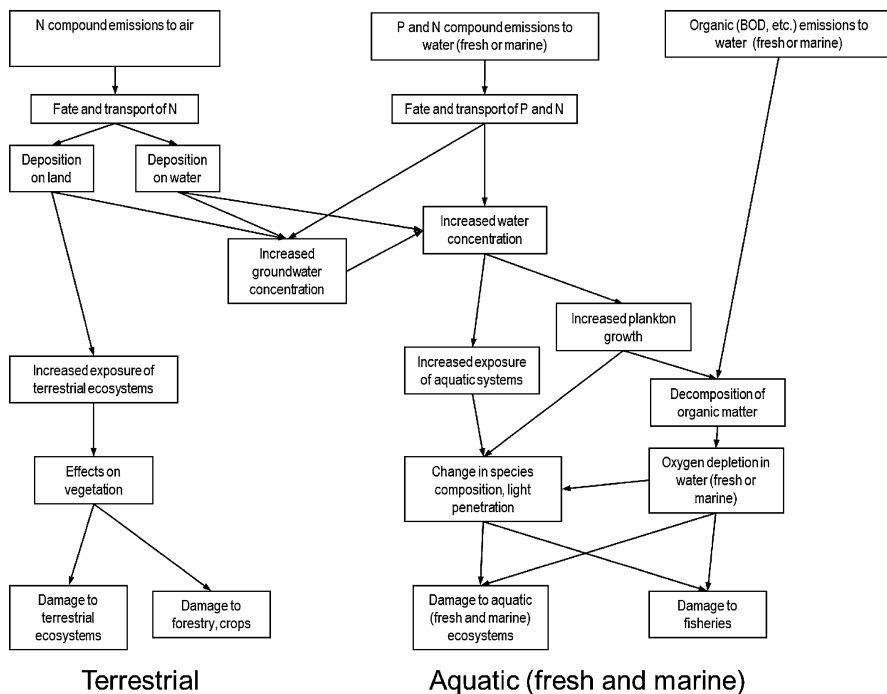
Fig. 10.1 Flow diagram of eutrophication impact pathway (Adapted from ILCD EC-JRC (2010b))

3.2 Aquatic Eutrophication

Increases of nitrogen and phosphorus, or inputs of respirable organic matter, in aquatic ecosystems can likewise change the structure and function of communities of plants and animals through a cause-effect chain that involves organisms in the benthos and water column. Excess nutrients first stimulate the growth of phytoplankton, increasing turbidity. This, in turn, affects plants in the light-dependent photic zone, as well as limiting predatory success of fish. The fish community shifts away from piscivorious toward zooplanktivorious species, removing zooplankton and favoring population increases in their prey, phytoplankton. Microbiological respiration of dead phytoplankton or other species (increased both due to direct nutrient input as well as the indirect ecosystem community changes) may lead to hypoxic conditions (i.e., low levels of dissolved oxygen in the water column), driving further changes (Kristensen and Hansen 1994). Such changes can also be caused by direct addition of excess organic matter (BOD or COD). To a certain degree, dissolved oxygen may be replenished via exchange with atmospheric oxygen at the lake surface.

However, some lakes undergo a seasonal stratification, limiting oxygen replenishment. The spring and summer warming of the surface layer, the epilimnion,

creates a sharp temperature gradient, limiting mixing with the lower layer, the hypolimnion. In this circumstance, decomposition of organic matter that has settled to the lake bottom consumes oxygen which is not replenished, potentially creating hypoxic or anaerobic conditions in the hypolimnion (Boehrer and Schultze 2008).

Low-nutrient water inputs through lakes, estuaries, or coastal zones, can serve to flush nutrients. Point-source emissions to well-mixed lakes can be modeled, but for diffuse nutrient sources in a watershed, the relationship between flow, nutrients, and eutrophication is less clear and can be difficult to model (Schindler 2006). The assimilation and movement of phosphorus through water bodies varies spatially and temporally (Withers and Jarvie 2008). Nitrogen, as it exists in multiple redox states, some of which are volatile, is more complex to model. Among other complicating factors, previously deposited P and N can be re-introduced to the water column during periods of anoxia (Levine et al. 1986; Mortimer 1942).

3.3   Freshwater and Marine Systems

It is important to distinguish fresh and marine waters; marine water bodies' salinity is not affected by freshwater inputs. Current understanding is that freshwaters are often—but not always—limited by phosphorus (Carpenter et al. 1998; Schindler 1977). Since the mid-twentieth century investigations that began to identify nutrients as drivers of eutrophication, it has been known that nitrogen may also limit freshwater productivity (Schindler 2006). Indeed, nitrogen, and even iron, may be co-limiting over time scales relevant to biological cycles, though it is still likely that phosphorus is the controlling, i.e. limiting, nutrient over scales of multiple years (Schindler 2006; Sterner 2008). If nitrogen is also limited in a water body, species capable of fixing N from the atmosphere (e.g., cyanobacteria) may be favored, reducing the extent to which N limits productivity (Smith 1983). In addition, N-limited lakes receiving anthropogenic nitrogen inputs may be eventually transformed to P-limited (Goldman 1988).

In contrast, productivity and eutrophication of marine waters has a more complex relationship with the two major nutrients than do freshwater systems, as hydrodynamics and trace elements play a more significant role (Grouzet et al. 2000). As a general rule, though, marine systems are typically N-limited (Jørgensen and Richardson 1996), and this provides a useful starting point for LCIA. While coastal zones and the upper layers of the ocean typically have greater biodiversity and productivity than other zones, zone-specific modelling of oceans has proven difficult. However, zones are taken into account in the cross-category biodiversity comparisons of the ReCiPe method (Goedkoop et al. 2009).

The limiting nutrient concept should be interpreted cautiously; as noted above, nutrient relationships in a water body can exhibit long-term changes, as well as change on sub-annual time scales. However, P limitation of freshwater and N limitation of marine systems provides a convenient foundation upon which to build LCIA models.

4 Contributing Substances (Classification)

As shown in Fig. 10.1, the major categories of emissions driving eutrophication are nutrients (nitrogen and phosphorus) and organic matter emitted to water. For aquatic effects, LCIA methods can differ in their approaches to capturing these ecosystem changes, with some endpoints focusing on nutrient enrichment, and others on oxygen depletion. Table 10.1 presents a summary of LCIA characterisation methods for eutrophication.

Consistent with the development of ecological understanding of eutrophication as driven by the supply of limiting nutrients, many LCIA methods primarily consider emissions of phosphorus and nitrogen compounds. Life cycle inventory (LCI) data may include a variety of forms of P and N; typical inventory substances are elemental P and N, phosphate (PO_4^{3-}), NH_3 or NH_4^+ (ammonia and ammonium), aqueous NO_3^- (nitrate), and gaseous nitrogen oxides (NO_x, representing the sum of NO and NO_2).

Since the addition of limiting nutrients may lead to oxygen depletion (due to microbial respiration of excess biomass), some LCIA approaches include substances that lead directly to oxygen demand, either biological (BOD) or chemical (COD) (see Fig. 10.1). The respiration of organic matter by bacteria consumes oxygen and forms the basis for the BOD metric; compounds not easily respired by microbial communities are captured in the COD.

LCIA methods that assess oxygen depletion, e.g., LIME (Itsubo and Inaba 2003), include organic matter (BOD and COD) as contributing substances. Although this biological material will contain some P and N, the mechanisms of action of these organic substances are distinct from those of the limiting nutrients, so double-counting is not a concern. The methods that do measure oxygen depletion therefore require an approach to translate P and N emissions to changes in oxygen levels.

5 Scale and Variability (Spatial and Temporal)

5.1 Scale

Atmospheric emissions of NO_x and NH_3 can be transported over a continental scale; e.g., the established RAINS and EMEP models for Europe (Alcamo et al. 1990; EEA 2009), or ASTRAP for North America (Shannon and Clark 1985; Toffoleto et al. 2007), affecting broad terrestrial regions, as well as being deposited on aquatic systems, though the latter is a minor source of aquatic nutrient inputs, given the small fraction of surface area occupied by freshwater relative to land.

In the case of freshwater emissions, the region of impact is defined by the downstream path of a receiving water body and the biogeochemical processes affecting nutrient transport. Some fractions of P emissions may travel ~1,000 km

Table 10.1 Summary table of LCIA eutrophication methods

Method	Impacted ecosystem	Midpoint	Endpoint	Unit (per kg emission)	Coverage (spatial, temporal)	Substances	Underlying model	Notes	Reference
CML 1992	Terrestrial, aquatic (combined)	Nutrification potential	–	PO_4^{3-}-eq.	Constant	N, NO, NO$_2$, NO$_x$, NO$_3$−, NH$_4^+$, P, PO$_4^{3-}$, COD	–	Potential based on redfield ratio	Heijungs et al. (1992)
CML 2002	Terrestrial, aquatic (combined)	Eutrophication potential	–	PO_4^{3-}-eq.	Generic	NH$_3$, NH$_4^+$, NO$_3^-$, HNO$_3$, N, NO$_2$, NO, NO$_x$, PO$_4^{3-}$, H$_3$PO$_4^{3-}$, P, P$_2$O$_5$, COD	–	European country factors available from Huijbregts and Seppälä (2000)	Guinée et al. (2002)
Eco Scarcity 2006		Ecopoints (EP) (assigned based on exceedance of critical load)		EP	Switzerland (+ regional waters)	Air: NO$_x$, NH$_3$; Water: N and P substances		Critical flow defined by political target	Frischknecht et al. (2009)
Eco-Indicator 99	Terrestrial		Affected target plants – probability of disappearance	PDF·m^2·year	Europe	NO$_x$, NH$_3$ (to air)	SMART2 (Kros et al. 1995)	Based on change in Dutch species (Latour et al. 1997), extrapolated to Europe. Acidification combined with eutrophication	Goedkoop and Spriensma (2000)
EDIP 97	Aquatic, terrestrial (aggregated)	Eutrophication potential		NO$_3^-$, N, or P-eq.;	generic	NO$_3^-$, NO$_2$, NO$_2^-$, NO$_x$, N$_2$O, NO, NH$_3$, CN-, total N, PO$_4^{3-}$, P$_2$O$_7^{2-}$, total P		factors can consider P or N-limited systems; uses Redfield ratio	Hauschild and Wenzel (1998), Wenzel et al. (1997)

Method	Aquatic/terrestrial	Eutrophication potential	Ecosystem area that becomes unprotected (UES)	Unit	Geographic scale	Substances	Model	Comments	Reference
EDIP 2003	Aquatic, terrestrial	Eutrophication potential	Ecosystem area that becomes unprotected (UES)	Aquatic: NO_3^-, N, or P-eq.; Terrestrial: m^2 UES (terrestrial)	Aquatic: European country; Terrestrial: Europe (44 regions or generic; 1990 or 2010 factors)	See EDIP 97	Aquatic: CARMEN; Terrestrial: RAINS	Aquatic adds fate factor to EDIP 97 site-generic factors	Potting and Hauschild (2005)
EPS	Aquatic		Fish and meat production (air emissions); Normalised EXtinction of species (NEX) (water emissions)	kg fish production; kg NEX	Global	To air: NO_x, N_2O, NH_3, N-tot; to water: BOD, COD, N-total, P-total		Convert BOD to O_2 demand using Lindfors et al. (1995)	Steen (1999a)
LIME	Marine	Dissolved oxygen concentration	Biodiversity, primary production, and economic value		Japan (specific bays modeled)	NO_x, NH_3		Limited documentation	Itsubo and Inaba (2003)
LUCAS	Aquatic (+ groundwater), terrestrial	Terrestrial: amount of emitted N that reaches ecosystem		kg N-eq.	Canada (15 ecozones)	N and P	ASTRAP (air), CARMEN (water)	Adapts EDIP 2003 to Canadian context	Toffoleto et al. (2007)
MEEuP	Aquatic	Eutrophication potential		PO_4-eq.	generic	N, NO_3^-, NH_4^+, P, PO_4^{3-}, P_2O_5, COD, BOD, SS, DOC, TOC NO_x (to water)	–	Substance weighting based on European directives (Vermeire et al. 1997, 2007)	Kemna et al. (2005)
ReCiPe	Freshwater and marine	Eutrophication potential	Species disappearance	Freshwater: P-eq. (mid), PDF·year·m^3 (end): Marine: N-eq. (mid)	Europe	N and P substances	CARMEN (water), EUTREND (air) (van Jaarsveld 1995)		Goedkoop et al. (2009)
TRACI	Aquatic	Eutrophication potential		kg N-eq. or kg PO_4^{3-} eq.	US (state resolution)	N and P substances, BOD, COD	ASTRAP (air), exorheic fraction (water)	Eutrophication potential based on Redfield ratio	Bare et al. (2003)

Impact 2002+ (Jolliet et al. 2003) uses terrestrial eutrophication from Eco-Indicator 99 and aquatic eutrophication from CML 2002

(Helmes et al. 2012). For marine emissions, the scale of impact is related to the extent to which receiving bodies create partially-enclosed systems with limited mixing with the larger ocean. Scales of impact can be quite large, e.g., zones of hypoxia up to 80,000 km^2 in the Baltic Sea (Diaz 2001; Hansson et al. 2009).

The most important spatial distinction is between P and N-limited water bodies. Stoichiometric (e.g., NH_3 vs. NO_3^-) and bioavailability differences between eutrophying compounds are minor relative to correctly capturing the limiting nutrient or possible spatial transport differences. The exclusion of the latter may diminish the relevance of an impact assessment (Potting and Blok 1994), and inclusion of spatial differentiation was identified as a major challenge for LCIA early on (Potting 2000). Finnveden and Potting (1999) had begun working on spatial eutrophication in the early 1990s (UNEP 2003). Some of the first published spatially-differentiated transport factors for eutrophication were country-specific estimates of marine deposition of air emissions of NH_3 and NO_x, and these were included in a multi-scale (country, Europe, world) assessment that included runoff and leaching from agricultural lands (Huijbregts et al. 2000; Huijbregts and Seppälä 2001).

Several studies have pointed to the possible variation, by orders of magnitude, in transport between sources and receptors. Transport differences between European countries for airborne nitrogen compounds can be up to three orders of magnitude (Posch et al. 2008; Potting et al. 1998b). At the impact level, also comparing European countries, the eutrophication potentials of nitrogen air emissions were found to vary by up to 1.5 orders of magnitude (up to 3.5 orders of magnitude for acidification) (Huijbregts et al. 2000). Differences in aquatic transport between European countries and US states, according to previous work, were less than one order of magnitude (Norris 2003; Potting and Hauschild 2005). However, for P transport, recent modelling has suggested possible variations of 3 orders of magnitude between US states (Helmes et al. 2012).

At small spatial scales, variation in the transport and eventual impacts of P and N can therefore be significant. On the aquatic side, emissions to rivers and lakes have different fates and impacts; this is a level of detail not captured in impact models nor inventory data. UNEP/SETAC working groups formalised the call for archetypical situations (Margni et al. 2008), which represent deviations from the default, generic situation and could be used when spatial differences cause a variation above some threshold factor (e.g., 2–10, depending on the study). Margni et al. (2008) recommended continental-level resolution as preliminary step. More nuanced divisions are also possible, provided that inventory data are sufficiently detailed: for freshwater transport, the presence or absence of large lakes downstream from emission sources is a possible archetypical division point (Helmes et al. 2012).

Comparing freshwater bodies or terrestrial areas, there can be differences driven by variation in climate, species composition, underlying geology, or previous environmental stresses, among others. Many of these factors control existing levels of nutrients; this may affect the limiting nutrient. However, within one nutrient limitation regime, the response of a water body to nutrient input may vary depending on existing levels of that nutrient. As a threshold concept, this presented challenges to developers of LCIA (e.g., UNEP (2003), for the existence of a cutoff

value can imply the existence of an infinite sink for a substance in the environment. Work by Struijs et al. (2010b) assumed a threshold of 0.3 mg/L as indicative of excess human nutrient inputs. An alternative approach has also differentiated response levels, but with impacts occurring at all existing P levels – i.e., without a threshold concentration (Payet 2006).

The time scales over which eutrophying emissions reach and cause impact in a receiving location vary by compound and the emission compartment. The time frame over which first-order impacts occur is generally rapid (i.e., proportional to nutrient uptake during the growing season), but second-order impacts are more complex and varied (e.g., remobilisation of previously sequestered nutrients). Atmospheric emissions of NO_x and NH_3 can be transported over a continental scale; however, they have atmospheric residence times on the order of hours to days (Galloway 2003). Once deposited to terrestrial systems, reactive nitrogen can persist for time scales ranging up to centuries in unmanaged forests (Galloway 2003).

For freshwater emissions, the hydrological cycle tends to move nutrients downstream relatively efficiently, with natural and man-made reservoirs delaying transport. For phosphorus, impacts happen throughout a river system; time scales can range up to years for areas that are upstream of large water bodies (Helmes et al. 2012). For nitrogen, impacts in the coastal zones are delayed while nitrogen is transported; this transport is largely tied to river transport, though wetlands may remove nitrogen prior to reaching marine systems (Galloway 2003).

There is often a time lag before substances emitted to groundwater may reach a down gradient fresh or marine water. In a model of European nutrients, Beusen et al. (1995) used a typical time scale of 50 years for nitrate emissions to groundwater. However, the majority of nitrate discharge was in groundwater for less than 5 years. Over longer time scales, one method to account for temporal variability is to include future emission scenarios (and corresponding environmental concentrations or sensitivities) (EC-JRC 2011).

5.2 Variability

As noted in Sect. 3, there are seasonal changes in eutrophication impacts. It is also possible for the limiting nutrient in a water body to change, both on sub-annual time scales or over a longer time frame (for example, if there is an increase in atmospheric N input). However, LCIA models have not accounted for this variability. In the former case, the added complexity would not necessarily contribute to enhanced life cycle impact modelling; in the latter, improved ecosystem models and loading data would be necessary.

At short time scales, eutrophication can exhibit strong variability. Areas at high latitudes may have substantial seasonal differences in available light, and thus times of year when microbial activity does and does not drive eutrophication. Many areas experience seasonal stratification (see Sect. 3.2), which hinders replenishment of dissolved oxygen in lake hypolimnion. The LCIA perspective does an adequate job

of capturing overall eutrophication impact trends at longer time scales. However, if the limiting nutrient changes over time, this would be imperative to capture in a model. Early LCIA models did not attempt to account for intra-annual variability (Udo de Haes et al. 2002).

Limnological science is not yet robust enough to understand the recovery of ecosystems after cessation of inputs (Schindler 2006), so using varying lengths of impact windows is not currently possible. However, it is clear that there are seasonal variations in eutrophication, e.g., for freshwater and marine systems (Conroy et al. 2010; Obenour et al. 2012); these subtleties pull LCIA characterisation towards models with higher temporal and spatial resolution, but the only models that capture these trends are highly parameterised, highly tailored models for specific systems (e.g., Obenour et al. 2012). Global models and the requisite input data for such models are not yet available.

6   Midpoint Methodologies

Eutrophication is a category with relatively few substances affecting the area of protection. At the effect level, assuming the critical distinction between P and N-limited conditions has been made, there are not large differences between such substances: from a nutrient supply perspective, the distinction between ammonia and nitrate is largely one of stoichiometry. Therefore, correctly modelling the fate and transport of eutrophying substances is critical. Those models that account for BOD and COD are able to tie in P or N inputs via assumptions about nutrients driving primary production and, hence, oxygen depletion.

Table 10.1 summarises currently available characterisation methods for eutrophication impact. The simplest approach, that of the initial LCIA models, is to assume a standard fate of eutrophying emissions, which can range from an implicit transport without losses to a fraction of emissions that reach terrestrial, freshwater, or marine ecosystems. CML 2002 (Guinée et al. 2002) and EDIP97 (Hauschild and Wenzel 1998; Wenzel et al. 1997) take the former approach for terrestrial eutrophication. Using the Redfield ratio (Redfield 1934), which is based on typical aquatic biomass stoichiometry, N and P substances are converted into phosphate or nitrate equivalents, representing an overall potential for eutrophication.

To varying degrees, all midpoint models make assumptions about or explicitly model fate and transport of eutrophying substances. At limited spatial scales, there may be minimal variation in fate and transport of airborne or aquatic emissions. As discussed in Sect. 5.1, the importance of spatial considerations has been established for emissions to air (Potting et al. 1998a) and freshwater. Therefore, refining transport models will be an active work area in future.

Another midpoint approach, used for terrestrial eutrophication, is to model the transport of substances, linking their environmental fate to deposition in sensitive areas. The accumulated exceedance model of Seppälä et al. (2006) uses the EMEP model for transport and a LRTAP critical load database (Posch et al. 1995) to

determine the assimilative capacity of the receiving area. EDIP 2003 expresses its eutrophication midpoint as an increase in unprotected area; this is a binary, on/off model, which maps airborne deposition predictions of the RAINS model (Alcamo et al. 1990) to the LRTAP critical load database (Potting and Hauschild 2005).

For aquatic eutrophication, methods may also employ fate models to calculate a nutrient enrichment. EDIP 2003 couples the RAINS model for deposition with CARMEN (Beusen et al. 1995; De Haan et al. 1996; Klepper et al. 1995) for aquatic fate; substances are converted to nitrate equivalents. The EUTREND model (van Jaarsveld 1995) is used in ReCiPe, which distinguishes freshwater as P-limited and marine systems as N-limited (Struijs et al. 2009). The method behind the ReCiPe factors also uses the CARMEN model, which incorporates soil, topography, and land use data at a grid scale of $1/6° \times 1/6°$, allowing calculation of factors for gross application of fertiliser to agricultural lands (Struijs et al. 2010a). The TRACI method uses a topological hydrological model (Fekete et al. 2002) to consider fate, at the U.S. state level, for emissions to air and water, converting all substances to phosphate or nitrogen equivalents, for freshwater and marine eutrophication, respectively (Norris 2003). TRACI also uses the Redfield ratio to relate N and P on a molar basis and to quantify a eutrophication midpoint (see Sect. 1).

An alternative approach is offered by the LIME method, which was developed specifically for marine waters in Japan (Itsubo and Inaba 2003). Nutrient inputs (N and P) and organic matter inputs, as BOD and COD, are linked to oxygen depletion in the hypolimnia of coastal bays.

Although there is still discussion regarding the treatment of organic matter, it is clear that making the distinction between freshwater and marine systems is essential. In future, as the supporting science grows stronger and models are more spatially and temporally explicit, there will also be considerations of multi-nutrient limitation paradigms. At present, however, differences in fate modelling assumptions are the main distinction at the midpoint level.

7   Endpoint Methodologies

Ecosystem quality can be expressed in a variety of ways that are currently challenging to measure (Curran et al. 2011). Energy, matter, and information flows are quantities for which measurement is, to some extent, possible (Goedkoop et al. 2009). Ecosystem information can be expressed at the ecosystem, species, or gene level. In LCIA, most endpoint methodologies use species as indicators of overall ecosystem quality. Most such approaches consider overall species, based on the observational record. It is challenging to estimate species distributions prior to human intervention.

Indeed, even the current distribution of species is complex, and modelling the extinction of species is challenging and would rarely realistically be attributable to the emissions associated with a single product or service, as would be modelled in LCIA. Therefore, it is more common to model the disappearance of species from a

given area over a given time, assuming that removing the environmental stressor would allow the species to re-colonise an affected area (Goedkoop et al. 2009). See Goedkoop et al. (2009) for a discussion of the weighting of species across terrestrial, freshwater, and marine environments.

Table 10.1 summarises currently available eutrophication impact methods. Some endpoint methods have adapted information available from regional or country-specific studies of species occurrence. For example, the Dutch Nature Planner (Latour et al. 1997) relates acidifying and eutrophying substances to threatened terrestrial species. This study is used in Eco-Indicator 99 (Goedkoop and Spriensma 2000) to characterise impacts in terms of a potentially disappeared fraction over an area and time (PDF·m^2·year). A Swedish study on the fraction of endangered species related to terrestrial and aquatic eutrophication is used in the EPS2000 method (Steen 1999a, b), which applies a generic fate assumption to N and P, as well as oxygen-depleting substances. One limitation of such approaches is the difficulty in extrapolating to other or larger areas, from continents to the globe. A study of macrofauna in Dutch waters was used to develop an effect factor relating P concentration to species sensitivity in the ReCiPe method, which is coupled to the midpoint fate model to calculate impacts as PDF·m^3·year (Struijs et al. 2009, 2010a). This model assumes no impact at P concentrations below 100 µg/L. A study of French freshwater invertebrates (Tachet et al. 2000) was used to create a model of species effects due to increasing P concentrations (Payet 2006); this was added to the IMPACT World + method (www.impactworldplus.org).

Considering the flow of resources from ecosystems, the LIME method (Itsubo and Inaba 2003) translates the oxygen depletion in marine ecosystems to reductions in biomass in the benthos as well as fishery catch decreases. These impacts are expressed in monetary units. Having been developed for specific locations, it is difficult to apply the LIME endpoints to other regions.

As the science to support endpoint modelling is generally not as robust as midpoint modelling (e.g., endpoint species sensitivity vs. midpoint fate and transport), these endpoint methods all carry higher uncertainty.

8   Recent Developments and Research Needs

The arc of developments for eutrophication continues forward, often following the themes of improving modelling of transport and effect, with respect to refined geographic resolution and increased geographic coverage.

Geographic specificity of fate models, such as the CARMEN model used by Struijs et al. (2010a), allows for more precise conversion of inventory (e.g., nutrient application to an agricultural field) into emissions to freshwater. Accurately modelling the fate of such emissions is an area of continuing work.

On the effect side, existing species sensitivity distribution (SSD) approaches are likely limited in their taxonomic coverage and geographic transferability. Extrapolating from taxa is difficult, as responses to stressors between taxa is weak (Wolters

et al. 2006), introducing uncertainty. Whether the underlying data source is Dutch (Struijs et al. 2010b) or French (Payet 2006), empirical relationships are constrained in application by the geographical specificity of their input data. SSD are not meant to be used in specific water bodies, nor can they be applied outside the region on which they were based without careful thought. These two impact methods are probably applicable in temperate climates, but use for (sub)tropical climates may be problematic. New work attempts to mine a richer dataset, adding analysis for freshwater eutrophication due to P at a global scale, comparing linear, average, and marginal effects, for autotrophs and heterotrophs (Azevedo et al. 2013b).

As understanding of ecological systems' responses to nutrient inputs improves, empirical and theoretical approaches to modelling eutrophication may change. Marine modelling of eutrophication impacts has yet to be formalised, and has been identified as a major challenge in this century (Cederwall and Elmgren 1990; Turner and Rabalais 2003).

LCIA has traditionally not dealt with combinatory (synergistic or antagonistic) effects of compounds (UNEP 2003), although work in ecotoxicity has explored this concept (van Zelm et al. 2007a). Future models should work to shed light on those situations where water bodies may not be not limited by single nutrients. Beyond nitrogen, iron may limit productivity, and silica can also play a role, via blooms of diatoms, in eutrophication (Schindler 2006; Sterner 2008); this should be addressed in future research. Such models will require understanding time scales of eutrophication; if a dynamic model were available, it could be used to create temporally variable impacts, as has been done for acidification (van Zelm et al. 2007b).

However, there is a need for a re-conceptualisation of underlying assumptions in current LCIA models for eutrophication. First, the choice of linear, average, or marginal effect model can significantly influence calculated effect factors (Azevedo et al. 2013a). Beyond the numerical values of effect factors, though, the meaning and goals of those factors should be considered: the marginal effect model discounts the importance of additional stresses to an already-degraded area, thus providing little impetus for remediation of degraded areas (Huijbregts et al. 2011). In contrast, an average effect factor based on a target level may serve societal goals for LCA, such as restoration or minimising further damage to eutrophied areas.

Secondly, the PDF category indicator for intrinsic ecosystem value assumes that species occurrence adequately represents biodiversity, which in turn is assumed to adequately represent a suite of ecosystem metrics (Struijs et al. 2009). There is a need to revisit assumptions of linearity of PDF with scale, and to consider the different implications of local, regional, and global loss of habitat. However, as Curran et al. (2011) point out, LCA captures composition of ecosystems, but there is a need to incorporate other metrics of biodiversity, including function and structure. For example, considering impacts on native, as opposed to all, species may be a useful area of investigation. Going beyond loss of habitat (which is implied with the PDF-area or PDF-volume approaches), LCIA could also consider a phylogenetic indicator as well as a metric that links species to ecosystems; the often-used but ill-defined term 'ecosystem services' may capture this idea (Curran et al. 2011).

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